EIFAC Occasional Paper No. 36 EIFAC/OP36

ISSN 0258-6096

EUROPEAN INLAND FISHERIES ADVISORY COMMISSION
INTERNATIONAL COUNCIL FOR THE EXPLORATION OF THE SEA

Report of the thirteenth session of the

JOINT EIFAC/ICES WORKING GROUP ON EELS

Copenhagen, Denmark, 28–31 August 2001

Copies of FAO publications can be requested from:
Sales and Marketing Group
Information Division
FAO
Viale delle Terme di Caracalla
00100 Rome, Italy
E-mail: publications-sales@fao.org
Fax: (+39) 06 57053360

EUROPEAN INLAND FISHERIES ADVISORY COMMISSION
INTERNATIONAL COUNCIL FOR THE EXPLORATION OF THE SEA

Report of the thirteenth session of the JOINT EIFAC/ICES WORKING GROUP ON EELS

Copenhagen, Denmark, 28–31 August 2001

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
Rome, 2003

ISBN 92-5-104941-6

PREPARATION OF THIS DOCUMENT

This report summarizes the presentations, discussions and recommendations of the Thirteenth Session of the Joint EIFAC/ICES Working Group on Eels, which took place in Copenhagen, Denmark, from 28 to 31 August 2001.

FAO European Inland Fisheries Advisory Commission; International Council for the Exploration of the Sea
Report of the thirteenth session of the Joint EIFAC/ICES Working Group on Eels. Copenhagen, Denmark, 28-31 August 2001.

EIFAC Occasional Paper. No. 36. Rome, FAO. 2003. 62pp.

ABSTRACT

The EIFAC/ICES Working Group on Eels met at ICES headquarters, from 28-31 August 2001, to finish the work initiated at its 1999 meeting on defining biological reference points for European eel management use. The review of available information revealed that the European eel stock is in decline and that fisheries is outside safe biological limits. Anthropogenic factors (exploitation, habitat loss, increased predation, contamination and transfer of parasites and diseases) as well as natural processes (climate change) have contributed to the decline. Latest recruitment data (spring 2001) indicated a further deterioration of the status of the stock. As management at local level has failed to address the global decline of the stock, the implementation of an international stock recovery plan is of utmost urgency. The Working Group recommended that an international commission for the management of the European eel stock be formed, to organize monitoring and research on eel stocks and fisheries, and to serve as a clearing house for regular exchange of information regarding landings and resource status.

Distribution:

ICES
Participants to the Working Group
FAO Fisheries Department
FAO Regional Fishery Officers
EIFAC Mailing List

CONTENTS

Page

1 INTRODUCTION

At the 87th Statutory Meeting of ICES (2000) and at the 21st meeting of the EIFAC in Budapest, Hungary, it was decided that:

The EIFAC/ICES Working Group on Eels {WGEEL} (Chair: W. Dekker, Netherlands) will meet at ICES Headquarters, from 28-31 August 2001 to finish the work initiated at its 1999 meeting on defining biological reference points for European eel management use. The Group should address the following terms of reference

- *In response to the 1998 EC request on providing escapement targets and other biological reference points on European eel for management use the Group should:*
  - *a) assess trends in recruitment and their causes and the effects on stock and yield of the species;*
  - *b) investigate the impact of fisheries on spawner escapement in selected systems;*
  - *c) define relevant units where escapement targets would be applicable;*
  - *d) where information warrants, propose preliminary biologically-based escapement goals for selected systems;*
- *propose management actions leading to the required escapement;*
- *report progress in work on improvements in the scientific basis for advice on management of European eel fisheries;* inter alia *on*
  - *a) development of harvest rate models for eel fisheries in data-rich systems;*
  - *b) assessment of density-dependent processes (growth and mortality) and their impact on spawner escapement;*
  - *c) development of reference points for management use in data-poor systems;*
  - *d) developments of procedures to verify effects of eel fisheries management measures, in data-rich and data-poor systems;*
  - *e) assessment of the (positive) impacts of management measures not directly related to exploitation, e.g. fish passes, habitat improvement, re-stocking, etc.*

Nineteen experts attended the meeting, representing ten countries. Additionally, ICES and EIFAC officers participated. The list of participants is given in the *Appendix*.

During the meeting of the Working Group, it was felt that ongoing management and research of eel necessitated consideration of some major issues that were not fully included in the Terms of Reference. It was decided not to exclude these items from the discussions and consequently this report also contains some discussion not directly related to the TORs. This applies in particular to management of eel stocks by measures other than regulation of exploitation, i.e. management of other anthropogenic impacts and by re-stocking.

The structure of the report essentially follows the Terms of Reference for the meeting, with additional sections on issues not related to exploitation inserted where appropriate.

2 TRENDS IN RECRUITMENT, FISHERY YIELD AND IMPACT FACTORS

2.1 Trends in recruitment

2.1.1 Recruitment data series

There are relatively few data sets which provide information on the recruitment of the European eel and these do not always adequately describe the size or pigment stages (glass eel or elver) of the recruitment material. Available time-series from 19 river catchments in 12 countries were examined for trends (Table 1). The data analysed were derived from both

fishery-dependent sources (i.e. catch records) and fishery-independent surveys across much of the geographic range of the European eel, and cover varying time intervals. Trends were examined over the entire duration for which data were available, in particular for the period after 1980, to investigate more recent possible changes. The sources of the data are clearly differentiated in Table I.

No upward trends were observed in any of these European data sets. Over the last two decades of all time-series, downward trends were evident, reflecting the rapid decrease after the high levels of the 1970s. Over the 1980s, the trend was downwards with the exception of the Erne in north-western Ireland in which no trend was apparent. In the 1990s most series have shown fairly stable low levels. The recent years show a continued decrease and the 2001 level is the lowest on record for all series where data has been reported.

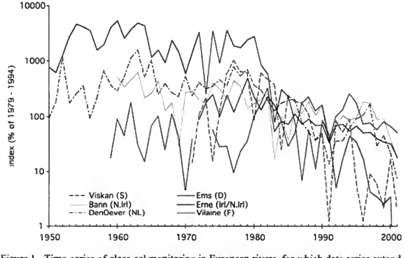

Figure 1. Time-series of glass-eel monitoring in European rivers, for which data series extend to 2001. Each series has been scaled to the 1979-1994 average.

2.1.2 Causes of the decline in recruitment

Several explanations have been put forward for the observed decline. As the timing and extent of the decline varies substantially – with a major decrease of recruitment in the Scandinavian area starting as early as in the forties and fifties whereas most continental monitoring stations see the largest decline in the eighties – it is unlikely that a single factor can explain everything. The present knowledge is not sufficient to decide between the alternative explanations and the following is a listing of hypotheses without any judgements of their relative merits.

A basic division is into anthropogenic and natural causes. For the latter the main hypothesis is a connection between the recruitment decline and a decadale scale change in the oceanic circulation (Castonguay *et al.*, 1994). The parallel decline of the recruitment of the American eel in some of it's distribution area, and the correlation between recruitment and the North Atlantic Oscillation Anomaly supports this model (ICES, 2001). Other natural causes that have been discussed are different diseases – viral infections or the swimbladder parasite *Anguillicola crassus*. This parasite spread rapidly in the European eel population in the early eighties. Infection causes swimbladder dysfunction and may impair the migration of mature eels. Predation by the greatly increased European populations of cormorants or other predators has also been discussed.

Fishing and habitat loss are the main anthropogenic factors. The impact of the eel fishery is discussed in Chapter 3.1. A large part of the European inland water habitat has been made inaccessible to eels by hydroelectric dams or other obstructions to upstream migration (Chapter 3.3). Even where recruits can pass in eel ladders, or are trapped and transported upstream, the loss of escapement can be substantial due to a high mortality when the silver eels pass through turbines during the downstream migration. In addition to the loss of habitat by obstructions, large areas of wetland have been lost through draining and land reclamation. Even very small streams and ponds are suitable yellow eel habitats. The total change of available inland areas for eel is unknown, the process has occurred gradually, mainly during the second half of the twentieth century.

The spread of environmental contaminants may contribute to the recruitment failure. The burden of persistent contaminants in inland and coastal waters has increased both in amount and number during the early period of decline, but during the more recent decline in the eighties the trend has been the reversed. Eels accumulate organochlorines and other fat soluble substances readily, and this may impair the migration and affect the survival of the larvae. A concern expressed more recently is the spread of endocrine disruptors.

2.2 Trends in stock and yield

2.2.1 Landings statistics

The Food and Agriculture Organization of the United Nations (FAO, Rome, Italy) maintains a database of fishing yields. Additionally, the International Council for the Exploration of the Sea ICES (Copenhagen, Denmark) maintains a database of landings of marine, Atlantic fishing yields. As the data in the ICES database exclude the major yield from the stock at forehand, i.e. the inland catches, preference was given to the FAO data.

Official landing statistics for many countries comprise only about half of the true catches in the 1980s and 1990s (ICES, 1988; Moriarty and Dekker, 1997), because of illegal and unreported catches, as well as lack of coverage of many areas in several countries. However, to some extend trends in the reported data will reflect true changes in fishing yields.

FAO eel landing statistics are presented in Table II and Figure 2. The data show a clear decrease of yield during the last 20 years in Denmark, Netherlands, Italy, France and Portugal. In Sweden, Germany, Spain and the United Kingdom a less pronounced decrease is observed. In Ireland a marked increase in catches has taken place possibly because the eel fishery was developed over this period. In Norway the catches seem to be stable.

The FAO catch return data do not necessarily reflect the status of the eel stock. Effort can be variable and underreporting the catches is a serious problem in most countries.

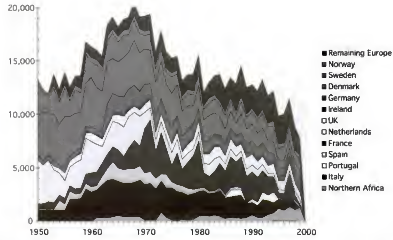

Figure 2. Landing statistics of the European eel in the past 50 years, as reported by FAO data base, with minor corrections.

2.2.2 Impact of recruitment decline on stock and yield

Impact on glass eel fisheries

In England and Wales, only hand-held dip nets are permitted for the capture of glass eels/elvers and fishing is concentrated in areas of high recruitment/easy capture, principally in estuaries of the River Severn and other rivers draining into the Bristol Channel. The number of licenses purchased per year was fairly constant at ~1 000 until 1994, but then rose to a peak at ~2 500 in 1997-98 as catch values increased due to demand for seed stock from new eel farms in China (Figure 3). Licence numbers subsequently declined to <1 500 in 2000 (data not included in Figure 3) as a result of farm-overproduction and imposition of import quotas by the Chinese. Provisional information suggests licence sales in 2001 were particularly low due to restrictions on access to fishing sites because of foot-and-mouth disease regulations.

There are few reliable catch records for eel in England and Wales. Catch returns are required from commercial licensees in some areas, but return rates are sometimes low. Commercial catches are commonly believed to be under-reported, as fishermen are reluctant to disclose such information due to perceived income tax implications. Catch data (available returns combined with estimates) have been collated by the Ministry of Agriculture, Fisheries and Food (MAFF) and reported annually (Figure 4). It should be noted that these data are considered to be very incomplete and of variable accuracy, both as a consequence of the factors outlined above and because assessment methods have varied between regions and from year to year.

As the great majority of eels caught in England and Wales are exported, estimates of the catch have also been possible from customs and excise export records (Knights *et al.*, 2001). Separating exports of recruits (glass eel and elver) from yellow/silver eels (not readily apparent from customs records) has necessitated estimating the quantities and total values of glass eels on the basis of their relatively much higher value per unit weight. Adjustments have

also had to be made to allow for imports/trans-shipments of glass eels. The customs and excise records have sometimes appeared to be incomplete or erroneous (especially in recent years following liberalization of inter-EU trade). Further complications have arisen from the fact that some exporters have not declared all shipments or the sources of eels. However, despite these caveats, the export data generally provide a reasonable match with the trends in catch data (Figure 4), as well as being in broad agreement with data on recruitment, catches and markets elsewhere in Europe.

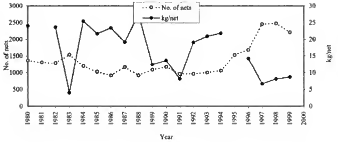

Figure 3. Glass eel fishing effort (no. of licensed nets) and CPUE (from export data) as kg/net, 1980-1999

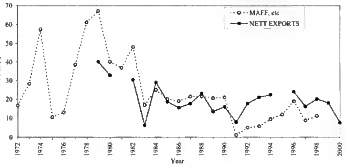

Figure 4. Glass eel catches (t) from MAFF/Environment Agency and export data, 1972-2000.

Impact on yellow eel fisheries

The quality of the eel catch data and the large and varying lag between arrival as glass-eels and recruitment to the fishery makes it difficult to demonstrate a causal link between recruitment and yield in the fisheries statistics. A case where long and relatively good quality data sets exist is the Swedish silver eel fishery in the Baltic (Svärdson, 1976). As an index for the recruitment to the Baltic the monitoring of recruits caught at an eel-ladder in a major river on the Swedish west coast (Göta Älv) has been used. The catch in the Baltic is dominated by females and a typical age at maturing is 15-25 years. Figure 5 shows the two data series. The decrease in recruitment clearly precedes the decrease in catch.

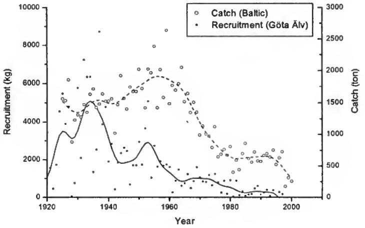

Figure 5. Time series of eel catches in the Swedish Baltic, including the Sound, and the recruitment immigration in a river on the Swedish west coast. The scattered points are annual values and the lines show 5 year FFT averages. No correction has been made for changes in fishing effort.

Stock in England and Wales

The decline in recruitment since 1980 has occurred almost all over the distribution area in synchrony (Dekker, 2000a). In the British Isles, however, the decline was much less pronounced. Additionally, the situation in England and Wales deviates from the remaining areas, in the sense that exploitation of the yellow eel stock is only marginal.

Recruitment in England and Wales has declined from peak values in the late 1970s, mirroring the changes seen elsewhere in Europe. Quantitative assessments of changes in eel stocks over the past 20 years have generally been hampered by a lack of robust time-series data. Surveys carried out in 1999 (Knights *et al.*, 2001) on three catchments (Rivers Frome,

Piddle and Dee) subject to low levels of exploitation, and for which reliable data from the 1970s and 1980s exist indicated the following:

- In the Frome and Piddle there has been a decline in biomass and in density, the decline being greater in terms of biomass. There appears to have been a decline in the number of glass eels entering the rivers as the number of eel < 150 mm is very low. The sex ratio in both rivers has changed from being previously male dominated to one where females now dominate the mature population.
- In the Dee, there was no indication of a significant change in either density or biomass, nor in the size structure of the population.

Examination of less robust data sets for a number of other rivers, indicated no statistically significant decline in stocks of yellow eels or changes in population structure over the last 20-30 years. However, the absence of widespread detectable changes in yellow eel standing crop/population structure should not lead to an assumption that recruitment is necessarily adequate, as in the majority of instances the programmes were not set up or designed to monitor change. In addition, given the relative longevity of eel, declining recruitment will have a delayed effect on the densities of eel in freshwater systems and the resulting spawner escapement. Thus the recent (and ongoing) decline in recruitment could lead to changes in the future.

2.3 Trends in restocking

Data were obtained from a number of countries, separately for glass eels/elvers and for bootlace eels. The size of 'bootlace eel' varies between countries. Most data available were on a weight basis. Weights were converted to numbers, using estimates of average individual weights of the eels stocked. These were 3.5 g for Denmark, 33 g for the Netherlands, 20 g for (eastern) Germany, and 50 g for Sweden. An overall number of 3 000 glass eels per kg was applied.

Recent time series available were available from (eastern) Germany, Netherlands, Sweden, Denmark and Northern Ireland. For Poland, an older time series was available. These are presented in Tables III and IV. In addition, some anecdotal information on re-stockings is available.

A downward trend in the level of re-stocking glass eels is observed since the early 1980s, down to 15 percent of former level (Figure 6). The level of re-stocking with bootlace eels has increased since then by 250 percent (Figure 7). The combined level of re-stocking (glass eels and bootlace eels) has decreased to 25 percent of the early 1980-level. Data from the Netherlands and Northern Ireland are available on the amount of intra-catchment re-stocking as compared to inter-catchment stocking (Table V). The percentage of intra-catchment stocking in the Netherlands increased in the 1990s to an average of 40.7 percent since 1990. In Northern Ireland the river Erne is re-stocked by intra-catchment transfers only. The percentage of intra-catchment re-stocking in Lough Neagh decreased from the early 1980s, to an average of 84.3 percent since 1990.

Re-stockings in the Republic of Ireland are dominated by the Shannon. Intra-catchment re-stocking of glass eels and bootlace eels in the river Shannon occur since 1958. The current average rate is 1.2 million recruits per year. The majority of these re-stocked eels are just pigmented.

The only data available for France are from the river Rhone. Barral (2001) estimates the total weight of re-stockings in the Rhone since 1978 at 22 000 kg (total over all years), 94 percent from intra-catchment stocking of bootlace eels (mainly 50-100 g) and the remainder

from glass eel stockings from the Atlantic. Probably these Rhone data are underestimated due to incomplete recordings.

Ciccotti (1997) describes the re-stockings in Italy, dating from centuries ago in the valli di Commacchio. In 1978-1982 re-stockings amounted to 0.4 million bootlace eel on average, imported from France. From 1990 onwards re-stocking practices have been abandoned. In the whole of Italy on average 35.2 million glass eels were stocked in 1988-1990. The origin of these glass eels is unknown. Additional stocking of 17.5 t of bootlace eels occurred but there is no information available on the sizes and numbers of these eels.

There are no current stockings in England and Wales. Historically these occurred on small scale, probably more than 15 years ago and intra-catchment.

Intra-catchment re-stocking of un-specified sizes of eels in Norway have been described for the Imsa river only during 1983-1996. These amounted to 0.2-13 kg and were lower than 1.0 kg since 1994.

Re-stocking of eels does not occur in Portugal and Spain.

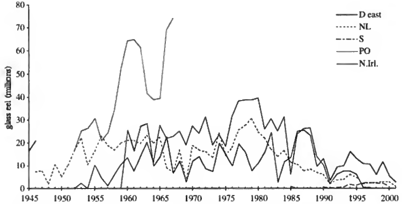

Figure 6. Re-stocking of glass eels in the Eastern part of Germany (D east), the Netherlands (NL), Sweden (S), Poland (PO, until 1967) and Northern Ireland (IR North).

There is no information on re-stocking available for Western Germany, Belgium, Finland, the remainder of the Baltic states, North-African states and states along the Eastern Mediterranean Sea.

Of all these countries where time series are lacking, probably only Italy, Ireland and the Western part of Germany are relevant for the totals. The Italian data in 1988-1990 show a level of re-stocking glass eels/elvers comparable with the cumulated data for Eastern Germany, the Netherlands, Northern Ireland, Denmark and Sweden. Applying the same trend in reduction of re-stocking to the Italian data, and considering that the current re-stockings of glass eels in the remaining countries probably do not exceed 5 million/year, this will give an estimate of the current re-stockings of ca. 30 million glass eels (10 tonnes) per year in Europe.

This is considerable less than the 33 tonnes mentioned by Moriarty and Dekker (1997). An unknown percentage of this amount concerns intra-catchment re-stocking but available data suggest that this may be substantial.

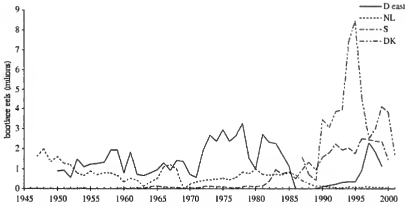

Figure 7. Numbers of bootlace eels re-stocked in the Eastern part of Germany (D east), the Netherlands (NL), Sweden (S) and Denmark (DK).

2.4 Trends in aquaculture

Aquaculture of the European eel ranges from highly industrialized, indoor facilities in northern Europe, through extensive culture in artificial ponds in southern Europe, to re-stocking of foreign glass eel in semi-natural outdoor waters for fisheries in northern Europe. All aquaculture fully depends on seed stock derived from the wild population, since artificial reproduction fails in the young larval stage. Additionally, aquaculture plants are used for quarantine of foreign glass eel to be re-stocked in outdoor waters (e.g. Sweden) and transports of half-products in-between aquaculture and fisheries occurs in and between countries (France, Italy). Obviously, the distinction between aquaculture and fisheries is hard to define.

For aquaculture production, no consistent long running time series exist. Data are available from FAO, from the Federation of European Aquaculture Producers, from previous meeting of the working group and from Kamstra (1999). An overview of the estimates is compiled in Table VI. In addition to the aquaculture in Europe, Eastern Asia (originally Japan, but recently predominantly China) has a large aquaculture industry, also culturing European eel.

Aquaculture of the European eel has started much later than the culture of the Japanese eel. In 1970, the European production was estimated at 3 400 tonnes, while the culture of the Japanese eel amounted 17 000 tonnes. In the early 1970s, European eels were cultured in Japan for a small number of years, with little result (Egusa, 1979). Since the mid-1980s Japanese culture of European eel has risen from 3 000 tonnes to 10 000 tonnes

nowadays. The European culture of the European eel is now estimated at 10 000 tonnes (Kamstra, 1999). This is to be compared to 40 000 tonnes of Japanese eel being cultured.

The aquaculture production in Europe is concentrated in Denmark, the Netherlands and Italy. The aquaculture in Denmark and the Netherlands is technically speaking highly developed and produces an increasing part of the total, while Italy has intensive as well as extensive culture systems, the latter with a declining production.

The landings from fisheries reported by FAO have declined from ca. 20 000 tonnes in 1970 to less than 10 000 tonnes nowadays (Section 2.2.1). This has coincided with a rise in European aquaculture production from almost nil in 1970 to 10 000 tonnes nowadays. This suggests that the total production in Europe has remained level. However, fisheries production is known to be almost twice the reported statistics, due to underreporting. Additionally, the rapid expansion of the East Asian production and consumption has resulted in the eel trade now being a global market, in which the apparently level European production is only one of the smaller constituents.

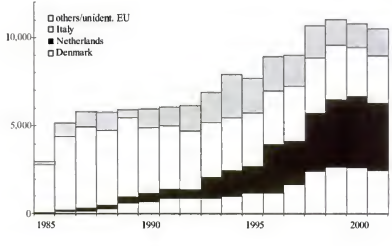

Figure 8. Trends in aquaculture production of the European eel.

3 ANTHROPOGENIC IMPACTS ON THE EEL STOCK

3.1 Impact of fisheries on spawner escapement

The maturing stages of eel have never been observed in the wild, but are undoubtedly purely oceanic in nature. Escapement of silver eel from the continent provides the best indicator of oceanic spawning stock biomass, but silver eel escaping the continental fisheries are probably more correctly defined as pre-spawners. There are no means available to assess potential losses between silver eel emigrating from freshwater and the oceanic spawning phase in the life cycle. Consequently, discussion will focus on the impact of fisheries on silver eel escapement.

The information available with regard to the impact of fisheries on silver eel escapement is very limited relative to the number of fisheries operating and there are few estimates of fishing mortality. Where available, mortality data have been provided, but these are inconsistent; both, instantaneous values and estimated losses over the freshwater phase have been included (see also Knights *et al.*, 2001). The data tend to be restricted to larger intensive fisheries that are not necessarily typical of the overall situation across the range of the species (Dekker, 2000a). These larger fisheries are geographically discrete, with the major glass eel fisheries mainly in the Biscay area and SW England, the freshwater yellow/silver eel fisheries concentrated in mainland Europe and Ireland, and fisheries in the Baltic focussing on silver eel more than elsewhere. The larger fisheries contribute only a small proportion of the total European catch (~5 percent, Dekker, 2000a). It is thus important to recognize that the following examples do not provide full insight into escapement processes of the species over its whole geographic range.

3.1.1 Impacts of emigrant silver eel fisheries on escapement

The most significant silver eel fisheries are based in the Baltic (using pound nets and similar passive devices) and in Lough Neagh, N. Ireland (using silver eel traps on the River Bann). A conservative estimate of overall escapement of silver eel from European waters is 553 tonnes (Moriarty and Dekker, 1997), but a Procrustean estimate based on all available evidence amounts to 1753 tonnes (Dekker, 2000c). Although there are no firm data, actual escapement is believed to be high in some river-based silver eel fisheries, due to inherent gear inefficiencies and current management actions to promote escapement (see below). In the Baltic, where silver eel dominates the catches, Wickström and Hamrin (1997) estimated using mark-recapture/mean recapture rates by commercial fishermen of between 35 and 49 percent, but could be as high as 69 to 76 percent. In the western Baltic, Pedersen and Dieperink (2000) reported recaptured rates of Carlin tagged silver eel in three pound net fisheries. Recapture rates varied between 19 and 38 percent and was dependent on the location and size of the fishery. The results indicate a high level of fishing mortality in the Baltic Sea which is further supported by the relatively short time interval between release and recapture of less than 16 days. These fisheries thus exploit the larger, more fecund, females. In addition, there are concerns about the ability of stocked eel to migrate successfully out of the Baltic.

The value of the above escapement estimates with regard to ensuring an adequate spawning stock biomass is restricted, due to low geographical coverage. It is impractical to set separate escapement targets for either sex, but it is emphasized that females being larger and older at migration are more vulnerable to capture (and turbine mortality in power stations) than males. Therefore, increased emphasis should be given to protecting females.

It seems reasonable to assume that density dependent processes do not operate in the oceanic migration towards the supposed spawning places. As such the percentage decline in silver eel escapement probably produces similar decreases in spawning stock biomass on the spawning grounds.

3.1.2 Impacts of yellow/silver eel fisheries on spawner escapement

The major yellow/silver eel fisheries are the Italian lagoons and the fyke net/long-line fisheries of the Netherlands, Germany, Denmark and N.Ireland. There are few reliable data sets with regard to effects on spawner escapement. The Italian lagoon fisheries are mainly closed and stocked systems and escapement can generally be regarded as zero. Estimates for the mortality rate in the IJsselmeer fishery are extremely high (F = 1.0; Dekker, 2000b) and spawner escapement is estimated to be low for males and practically nil for females. Exploitation of yellow eel in Lough Neagh is also high and, although not quantified, escapement of silver eel is assumed to be in the region of 20-25 percent of the yield of the

fishery. These two quantified larger scale fisheries (IJsselmeer and Lough Neagh) constitute a notable part of the total yellow/silver eel fisheries. They cannot be considered to be representative of the multitude of smaller fisheries that make up the rest (Dekker, 2000a), due to highly variable levels of exploitation. Consequently, accurate assessment of the continent-wide escapement is unachievable (Dekker, 2000b). Along the west coast of Sweden pound net and fyke net fisheries can be relatively efficient, catching more than 96 percent of the stock between 370 and 650 mm (Svedäng, 1999).

The above examples illustrate that yellow eel fisheries can impact upon silver eel escapement. The effect of other yellow eel fisheries on spawner escapement is unknown. However, assessment of lake IJsselmeer fisheries (Dekker, 2000c) suggests that even moderate exploitation of yellow eel results in substantial reduction in silver eel production. Dekker (2000c) showed that if a yellow eel fishery was operating at F_{max} (in the case of the IJsselmeer the minimum legal size is 280 mm) then the spawning escapement would be reduced to 10 percent of the unexploited biomass. As such, controlling yellow eel fisheries below their local optimum might be crucial for sustained exploitation of the stock.

3.1.3 Impacts of glass eel fisheries on spawner escapement

The reduction in egg deposition as a result of glass eel exploitation will be equivalent to the exploitation rate only if there is no density dependent change in sex ratio, growth, survival or emigration rate of the subsequent life stages. This, however, is not assumed to be the case.

The major glass eel fisheries are based on estuaries facing the Atlantic coast of France, Spain, Portugal and south-west England (Bristol Channel). The major market in the past was for direct human consumption in Spain and Portugal, plus stocking in central, northern and eastern Europe. However, high demands as a seed source for aquaculture in Europe (Italy, Netherlands and Denmark) and, in particular, the Far East, have recently resulted in very high prices. Effectively, none of the glass eel catches used in aquaculture yield spawners. Limited re-stocking of cultured eel has been initiated in some parts of Europe, but this appears to be mostly for the benefit of fisheries and is assumed to produce relatively few spawners.

There is little information on the impact of glass eel fisheries on recruitment into freshwater or the subsequent escapement of silver eel, for selected systems. However, natural mortality (exacerbated by density-dependent factors) is expected to be very high where abundance is very high in relation to the carrying capacity of the receiving river. Glass eel runs that exceed a river's carrying capacity may be a source for transfer to other rivers (but may also be important in contributing to the overall ecosystem). In the River Severn, Knights *et al.* (2001) compared the density of yellow eel in 1998/1999 with that found in 1983. The authors concluded that there was no substantive evidence for a major change in eel density or biomass over the time period even though there had been a major decline in recruitment of glass eel to the system since the early 1980s. The authors did, however, report a significant reduction (ca. 50 percent) in the proportion of eel <150 mm in sites from the lower Severn between the two time periods. Conclusion of impact or lack of impact must be made with caution as comparison between the surveys in 1998 and 1999 indicated large temporal variability (ca. 30 percent) in density and biomass.

In France, very high levels of fishing mortality have been recorded in certain glass eel fisheries. For example, it ranges from 20-25 percent in open estuaries such as the Adour, to 98 percent in closed estuaries such as the Vilaine (Briand *et al.*, 2000a). On the River Loire, a model showed that a decrease in glass eel recruitment is followed by a reduction of yellow eel population biomass 8 to 15 years later (Feunteun *et al.*, 2000a). This is consistent with an observed reduction in subsequent silver eel catches (Boisneau and Mennesson-Boisneau,

2001). In another example, on an obstructed catchment (River Vilaine, Brittany) which has very high fishing effort on glass eel, a reduction from 99.6 percent to 96 percent in fishing mortality and the use of a fish ladder may have resulted in an increase (2.8-fold) in the density of yellow eel in the watershed (Briand *et al*., in press, a). The migration and settlement pattern in the lower reaches of the river would suggest that this section of the catchment was at carrying capacity.

3.1.4 Impact estimates

Information is required on the relative impacts of fisheries compared to natural causes of mortality at different stages of the life cycle in order to help clarify whether stocks are being over-exploited or whether fishery control measures are required. Estimates of natural and fishing mortality on a case by case basis are presented in Table VII.

On the basis of this limited available information it is evident that fishing mortality can equal or exceed natural mortality. A brief qualitative assessment of the impact of fisheries in the various countries is presented in Table VIII. This table summarizes the educated opinions. Impact was classified into three categories; low where fishing mortality was equal to or less than natural mortality ($F \leq M$), optimal where the maximum yield per recruit was being obtained from the fishery (F_{max}) and over-exploited where growth-overfishing will be evident ($F \geq 0.1$). In the latter case, the spawning escapement is considered to be minimal. In some cases, however, depending on the legal size limit of the fishery, operating at F_{max} can reduce the spawning escapement to 10 percent of the unexploited biomass (Dekker, 2000b).

Overall, yellow eel exploitation is low in areas where fisheries predominantly target glass eel (England and Wales, France and Iberian Peninsula). Elsewhere, fisheries are most often optimized for yield.

3.1.5 Conclusion on impact of fisheries

It is impossible to assess the effect of fisheries on the overall escapement of the European eel stock with any real confidence as there are insufficient data and existing estimates for specific fisheries are mostly rather crude. Hence, stock-wide management targets can not be derived. However, the available information indicates that fisheries on all life-stages *can* and often *does* impact upon spawner escapement within particular locations and further suggests that some fisheries are capable of completely precluding escapement of potential spawners from a catchment or fishery. It follows that further controls on local fisheries on all components of the stock are appropriate and should contribute to the overall enhancement of production and escapement of spawners.

3.2 Effects of transfers and re-stocking of eel

3.2.1 Magnitude of re-stocking

Re-stocking of eel has a long tradition, in some countries going back to the nineteenth century or earlier. It has been practised in nearly all EU-states, several middle and eastern European states, in northern African states and Norway. Due to the increasing prices, higher demands for aquaculture and lower catches, the re-stocking of inland waters in Europe with glass eels has dropped to ca. 15 percent of the early 1980s level. This is equivalent to 5 percent of the most recent estimate of the total glass eel catch (583 tonnes per annum, Moriarty and Dekker, 1997). Although the re-stocking of bootlace eels increased since the early 1980s, the combined numbers re-stocked (glass eels and bootlace eels) decreased to 25 percent of the early 1980s.

The amount of glass eel used for re-stocking (both from inter-catchment and intra-catchment transports) may exceed the natural recruitment in some of the glass eel importing

countries (Dekker, 2000b). Intra-catchment re-stockings are transfers within a catchment. Inter-catchment stockings are transfers between catchments.

3.2.2 Re-stocking and local populations

Both transports within and between river systems have occurred. Transports within river systems consist of estuarine glass eel fisheries for stocking up-river areas (e.g. glass eel in the river Bann being re-stocked in Lough Neagh). Transports between river systems involves re-distribution of glass eel over rivers within countries (e.g. Swedish west coast catches being re-stocked at the east coast) as well as long-distance international transports from the Bay of Biscay and the Bristol Channel to northern and eastern Europe (e.g. development of eastern European fisheries outside the natural distribution area).

Knights and White (1998) have described effects of eel transfers on local eel populations. Experiments show reduced growth at higher densities, but it is unknown whether these densities are reached in re-stocking practices. Re-stocking, therefore, might induce a decrease in growth rate in recipient populations if carrying capacity is exceeded. There are indications for differences in growth performance of glass eel originating from geographically different regions (Klein Breteler, 1994).

Fisheries, handling and transport of glass eel for re-stocking generate a mortality of unknown magnitude. Little information is available about mortalities of glass eel in the estuaries in the traditional donor areas. Mortality of ascending eel has been shown to be density-dependent within a year-class, but densities comparable in magnitude to the natural recruitment in the Bay of Biscay have not been assessed. Glass eel densities in donor areas are probably so high that most glass eel will die naturally when not fished; re-stocking will undoubtedly improve overall survival of the recruitment material in any year (EIFAC/ICES, 2001).

In several cases an increase in relative numbers of male eel has occurred following re-stocking. Higher densities of eel also seem to be related to a dominance of males. Transfer of glass eel from areas of high eel density to areas of low eel density may promote the overall production of females.

Transfers of eel, for trade and for re-stocking purposes, present risks of spreading diseases and parasites. *Anguillicola crassus*, a swimbladder parasite, has invaded wild eel populations throughout Europe after unintended introduction from the Far East. Negative effects of this parasite on local eel populations have been reported. Risks of transfers of diseases or parasites apply particularly to transfers of eel between catchments but, to a lesser extent, also apply to transfers within catchments.

3.2.3 Re-stocking and spawning stock

Homing of silver eel derived from transfers

Re-stocking of glass eel and/or bootlace eel increases local eel stocks and might eventually result in higher escapement of silver eel. However, there are indications that silver eel derived from re-stocked French glass eel show a reduced ability to successfully navigate their way out of the Baltic Sea on their spawning migration (Westin, 1998). The contribution of re-stocked eel has therefore been questioned by EIFAC/ICES (2001).

Genetic Considerations

Recently available genetic evidence does not support the long established view that a single spawning stock breeds panmictically in the ocean (Wirth and Bernatchez, 2001).

Results from genetic studies suggest that three putative, genetically distinct sub-groups may exist:

- *Northern European* - corresponding to the Icelandic stocks
- *Western European* - including Mediterranean, western European and Baltic stocks
- *Southern European* - corresponding to eel stocks of Morocco.

If natural gene flow between the putative sub-groups is high, the risks associated with transfer and re-stocking is low. Data on gene flow are not available.

Contributions to the escapement

Contributions of re-stocking to the escapement of eels depend on mortalities, both natural and by fisheries, mortalities by hydropower stations included. There are no case studies available in which mortalities of both re-stocked and not-re-stocked eels from the same original population are compared. Generally the re-stockings occur in more upstream and more isolated waters as compared to the downstream catch places, giving more and better opportunities for the fisheries and hence, possibly, higher mortalities. Higher growth rates of the re-stocked eels may occur due to lower densities and, therefore, lower natural mortalities may counterbalance possible higher fisheries mortalities. The quantitative net effect is unknown and will largely depend on local factors.

3.2.4 Re-stocking and fisheries

Re-stocking can be a cost-effective means of restoring or maintaining yields in fisheries (Knights and White, 1998). To this end, it is essential in catchments with barriers where fish passes are ineffective and in isolated waters suitable for eel. Stocking in the Baltic and in Central-European countries occurs mainly because of shortages in recruitment and reflects the unequal distribution of recruitment material across the range of the European eel.

Lough Erne fishery is completely dependent on re-stocking, and glass eel re-stocking contributes to the Lough Neagh fishery. Eel fisheries in the inland waters of the Baltic countries (specifically Poland and the eastern part of Germany) depend almost completely on re-stockings. Dutch fisheries (excluding lake IJsselmeer) also rely upon re-stockings. Yield per re-stocked recruit (glass eel) ranges from 20 to 90 g in the Baltic, but figures for Lough Neagh and Lough Erne fisheries are substantially lower.

3.2.5 Re-stocking and other components of ecosystems

Introductions and re-stocking of eel can affect the abundance of the crayfish *Astacus astacus* and possibly also signal crayfish *Pacifastacus leniusculus* when the stock density is high and the crayfish are under ecological pressure. Impacts on other common crayfish species, such as *Austropotamobius pallipes* have not been assessed. There are no known effects on native local fish populations, except where densities of eel are high or under extreme environmental conditions.

3.2.6 Re-stocking in obstructed water systems

Regulation of natural river systems has obstructed migration routes of eel in many catchments. In these cases, re-stocking of glass eel derived from down-stream sources restores former local natural conditions and may contribute to the escapements. Of the total of 87 335 km^2 of continental waters, 3.6 percent have been classified as 'artificially obstructed' (Moriarty and Dekker, 1997). Hydropower stations reduce the chance of successful emigration of silver eel, in particular for the larger female eel. Eel ladders, downstream migration facilities for silver eel and intra-catchment re-stocking should be considered for restoration of eel stocks in obstructed waterways.

3.2.7 Conclusions on re-stocking

- The magnitude of the current re-stockings of glass eels is about 5 percent of the total recruitment of glass eel on the European coast. Part of it concerns intra-catchment re-stockings. Most bootlace eel re-stockings are intra-catchment re-stockings.
- The current re-stockings generally contribute to the fisheries.
- It is unknown to which extent re-stockings contribute to the escapement of silver eel.
- Escapees from long-range re-stockings may not be able to find their spawning places, but there are no indications of genetical impacts, provided that no Moroccan or Icelandic eels are used.

3.3 Impact of habitat loss on the stock

Fisheries for eel are wide spread and the assessment of the impact of exploitation has a long tradition. Clearly, management of eel fisheries is an essential part of a management plan. However, sustainable management and a stock recovery plan should also take into account anthropogenic impacts upon the stock, other than exploitation. The decline of the eel in Europe is often related to the decline of its continental habitat, its accessibility and its quality. The relative magnitude of these factors, in relation to the impact of exploitation, has not been quantified, but it seems likely to be significant in many European countries.

Among 69 European rivers studied to establish a new EU Water and Wetland index, only five (in Finland, Scotland, Wales and UK) were considered to be almost pristine and 50 were of poor quality due to impacts of canalization, pollution and altered flow regime (www.panda.org/europe/freshwater/wwi). Good ecological status as required by the EU Water Framework Directive currently is only met in the upper reaches of the 14 largest rivers in Europe, amongst which are: the Rhône, the Seine and the Loire (France). However, the situation is probably underestimated because most countries have inadequate environmental monitoring systems to safeguard their water resources. According to the World-Wide Fund for Nature (WWF) this is particularly the case in Belgium (Wallonia), France, Greece, Northern Ireland, Scotland, Spain and Turkey.

The loss of habitat is assumed to have strongly effected the capacity of the inland aquatic habitats to produce eels. Habitat destruction, loss of upstream accessibility, troubled escapement through turbines and deteriorated water quality all have had negative effects on local eel stocks.

3.3.1 Upstream accessibility of habitats

Loss of freshwater habitat due to construction of dams has occurred more in southern European countries than in northern countries (Moriarty and Dekker, 1997), reducing the potential eel production in these regions. On the Iberian peninsula, 70 percent of Portuguese and 93 percent of Spanish river habitats have restricted access for glass eel due to human intervention (dams without fish passages or contamination), including the largest river catchments (Ebro, Duoro and Tagus). Artificially restricted habitats not accessible to eels, mainly due to dams, include 50 percent of the Italian, 10 percent of the French, 5 percent of the Irish and 20 percent of the German rivers. Man-made impassable obstructions are almost absent in Great Britain and Sweden, dams being compensated with fish passages or upstream transportation.

The timing of construction of large dams in Europe coincided with the decline of European eel populations (Figure 9), with the largest number of dams built in southern countries (Spain, Turkey, Italy and France). The construction of new dams and water

diversions could further reduce the availability of river habitats to eels unless they are supplied with proper eel passes. The installation of effective eel passes at dams could reclaim eel habitats from upstream catchments. The installation of a fish passage at the dam in the River Vilaine (Brittany, France) has resulted in fast colonization of the newly accessible habitat by naturally recruiting glass eels and an 2.8-fold increase in density of yellow eels (Briand and Fatin, 1999).

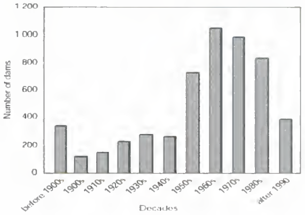

Figure 9. Number of large dams commissioned by decade in Europe. Data from ICOLD World Register of Dams (http://www.dams.org). Data for after 1990 is under-reported.

Impact of major dams is reported to restrict accessibility to upper reaches, but dams also modify habitat quality, transforming basically running shallow waters into deep still and eutrophic habitats. However, many major dams are not totally impassable as small eels may pass over wet vertical walls (Legault, 1994). Small dams, or hydraulic equipment such as bridges, pipes under roads, small weirs for irrigation, etc. often create temporary obstructions, and create a delay in migratory movements. Such conditions are assumed to increase mortality resulting in reduced eel stocks (Feunteun *et al.*, 1998). Impacts of obstructions have been reported for the Severn (UK) (Aprahamian, 1988; White and Knights, 1994, 1997a, 1997b), Thames (Naismith and Knights, 1988, 1993) and various smaller rivers in the United Kingdom (Turnpenny, 1989; Mann, 1995). A number of fish passes have been installed during recent years to facilitate the movements of eel and other riverine fish, but little has been done to monitor the efficacy of these structures. In the UK, three major tidal barrages have been constructed recently: at the mouths of the Rivers Tawe and Taff (S. Wales) and in the estuary of the River Tees (N.E. England). It is too early to say to what extent these obstructions may impact on the eel populations of these rivers.

There is currently no European synthesis on the number and location of minor and major dams and obstructions on river systems, but some regional examples might illustrate the impact of considerable river development and related impacts on upstream migrations of eels.

In France, in the River Rhone (100 000 km^2), there is about one hydrodam every 30 km on the main river. This has resulted in very low densities of eels in the river and no eels in most of the tributaries. Intensive re-stocking was conducted since the late seventies to sustain the riverine fisheries (Barral, 2001). Similar conditions are reported in the Seine catchment (89 000 km^2)and in international rivers such as rivers Rhine or Meuse.

Some large river systems such as the Loire (120 000 km^2) or the Gironde (100 000 km^2) are still not obstructed (about 300 km and 200 km from the tidal limit, respectively). However, the tributaries are highly developed for pleasure or commercial navigation and for flood control. For example, in the River Maine, a tributary of the Loire, there is about one dam or weir every 2 to 3 km (Feunteun *et al.*, 2000b).

In downstream stretches (<20 km of the tidal limit) of 28 minor rivers of the French Mediterranean coast (including Corsica) a total of 62 dams were listed. A total of 66 percent of the hydraulic constructions reduce severely upstream movements, only 1.6 percent had efficient eel ladders (Barral, 2001). On the Atlantic, Channel and North Sea coasts, a high percentage of minor rivers are obstructed between the estuaries and upstream reaches.

Spain has the highest number of dams in Europe, with approximately 1 200 operating large dams. Fish passes exist at only 15 percent of dams in large rivers (Nicola, Elvira and Almodóvar, 1996). The existing fish passages are mainly concentrated in the North of the country and are designed for Atlantic salmon. In general, most Spanish fish passages are old, inefficient or non operative, making most dams effective barriers to upstream eel migrations (Nicola *et al.*, 1996). Portugal has also been affected by the construction of dams starting in the 1950's, and presently there are over 100 large obstructions and many more small dams. The existing fish passages are generally ineffective, poorly designed or non-functional and are the main reason for the disappearance of several migrating fish species (Valente, 1993).

In conclusion, most European river systems are highly obstructed by dams, weirs, and other constructions.

3.3.2 Destruction of habitat

During the 20th century, most European rivers and aquatic systems have been reconstructed intensively. Reconstruction policies were mainly aimed at developing agriculture, navigation, industrial and urban areas. The most affected areas are wetlands and secondary river channels which were subjected to destruction by either reclamation or dredging practices. Overall wetland losses exceeding 50 percent of original area have been reported by the Netherlands, Germany Spain, Greece, Italy, France and parts of Portugal (Jones and Hughes, 1993). Considering density dependant mortality, growth and emigration, the loss of wetlands is assumed to have reduced the available eel habitats in Europe by at least 50 percent. Currently, the habitat area is estimated at over 87 000 km^2 (Moriarty and Dekker, 1997).

A few case studies might illustrate the magnitude of the problem. Many major river systems in Europe, including the rivers Rhône, Rhine, Meuse and Seine, were heavily reconstructed between the fifties and the seventies to favour commercial navigation, hydropower electricity generation and flood prevention. As a result, most of the wetlands were drained and the secondary channels destroyed.

In the Rhône river, the flood plains and secondary river channels between Lyon and the delta (approximately 350 km) originally extended over an area of 1 to 3 km width. They have progressively been reduced to a 300 m wide channel with low habitat suitability for eels, resulting in a loss of 200 to 500 km^2 of freshwater habitats. In the Loire river, dredging has lowered the main channel by one to two meters over 200 km in the downstream reaches. Consequently, the secondary channels are now flooded only during a few days each year, resulting in a loss of 100 to 200 km2 of suitable habitats.

Coastal freshwater marshes of France cover about 250 km^2 of which 10 percent are aquatic habitats with a dense population of eels (50-150 g/m^2) (Feunteun *et al.*, 1999). These have progressively been destroyed to favour agriculture but nowadays they are often abandoned. During the past decades, this situation was responsible for a rapid decline of the water surface by about 50 percent which now covers an area of about 10 km^2 (Feunteun *et al.*, 1999).

If we only consider these three examples, it can be assumed that in France alone at least 300 to 700 km^2 of highly suitable habitats for eel have disappeared representing 12 to 28 percent of the 2 500 km^2 of freshwater bodies actually available. (Moriarty and Dekker, 1997).

Loss of habitat did result in a decrease of the eel stocks and a corresponding reduction in silver eel escapement. Therefore, habitat destruction must be considered as one of the major causes for the decline of European eel stock.

3.3.3 Downstream migration

Mortality caused by hydroelectric turbines is well documented. Direct mortality ranges between 0 and 100 percent according to site characteristics, generator system design and turbine management procedures. For example, Knösche, Zahn and Borkmann (2000) show that in large systems average turbine eel mortality is 28 percent. Many large rivers have a series of dams (up to 14 in the Rhône river). Therefore silver eels leaving downstream areas are exposed to the turbine mortality several times during their downstream migration. Therefore, even low mortalities at individual dams will result in high overall mortality rates for emigrating silver eel.

Water reservoirs also have an impact upon silver eel escapement. Most of the dams on reservoirs have not been designed to enable downstream migration. Passage through dams is often only possible through bypass tube systems, designed to produce minimum discharge releases. Induced mortality by bypass tube outlets as shown to be about 100 percent in a small river system of northern Brittany (Legault *et al.*, in press).

Large dams are also assumed to delay downstream runs for up to several months, until maximal flooding conditions occur and overflowing of the dams occurs. The consequence of delays in downstream migration on the breeding success of eels is unknown.

In conclusion, obstruction to downstream migration and mortality caused by turbines are assumed to reduce silver eel escapement considerably. The overall impact is probably in the same order of magnitude as that of exploitation.

3.3.4 Water quality, contamination and breeding sueeess

Reduced and deteriorating water quality has been reported in water systems all over Europe. Due to the improvement of management policies, in the past decade concentration of contaminants has decreased in many large river systems of Europe.

Contamination does rarely induce direct mortality in eels (Knights, 1997). However, a recent review (Robinet and Feunteun, 2002) shows that contamination, even at very low level, by PCBs, dioxin and organophosphorous pesticides, result in an inability to store lipids or a premature silvering. A number of lipophilic persistent contaminants are also suspected to be released in oocytes during maturation of females creating egg and larval mortality.

4 RELEVANT GEOGRAPHIC UNITS FOR MANAGEMENT OF EEL STOCKS AND FISHERIES

Management options discussed previously and below refer to whole-stock conservation limits which need to be translated into appropriate local-system targets. The European eel population shows limited genetic variation only at large geographical scales (Maes and Volckaert, 1999; Avise *et al.*, 1986; Avise *et al.*, 1990), but other characteristics of the stock vary at distances of few kilometres (Dekker, 2000a). Moreover, fisheries are generally organized at small to very small scale (Dekker, 2000a), with very little and mostly clinal geographical differentiation (Moriarty and Dekker, 1997). Neither biological characteristics of the stock nor structure in exploitation patterns provide a key to develop relevant geographical management units at reasonable scales. *The Working Group felt that management of eel should ideally occur primarily on a catchment by catchment basis.* The catchment unit should include all fisheries and other anthropogenic impacts that occur on an eel stock and should also assist in the maintenance of genetically distinct populations in the event that the species was found not to be panmictic. The catchment approach poses two difficulties for implementation:

1. many small watersheds exist for which no information on eel is available to fisheries managers, and
2. in large watersheds (e.g., the Rhine) several fisheries management jurisdictions are involved in the management of one eel stock.

It is therefore recommended to focus on jurisdictional entities (countries, regions, etc.), allowing for differentiation by life stage and by catchment area. This will entail:

- deriving appropriate targets from the best available catch (or effort) statistics and units of measurement available (e.g. see summaries in Moriarty and Dekker, 1997 and this report, Chapter 2);
- setting, applying and enforcing targets as appropriate throughout the jurisdictional area of fishery controls to achieve the overall limits recommended above;
- in areas where during their life cycle eels migrate through several jurisdictions, co-operation between the jurisdictions involved to meet the management objectives for this eel stock.

5 PRELIMINARY ESCAPEMENT TARGETS

5.1 Biological reference points and the precautionary approach

ICES has recognized that a precautionary approach should be applied to fishery management and that reference points are a key concept in its implementation (ICES, 2000). These reference points could be stated in terms of fishing mortality rates or biomass with the intention of ensuring that the stocks and their exploitation remain within safe biological limits. Implicit in the development of reference points is the assumption that there is a relationship between spawning stock and recruitment. The precautionary approach dictates

that unless it can be scientifically demonstrated otherwise, such a relationship between stock and recruitment should be assumed to exist (ICES, 1997).

5.1.1 Reference points

The value of establishing reference points depends on the consequences to the resource of variations in spawning stock abundance. There are two hypotheses to consider in deciding whether spawning stock reference points are appropriate for the European eel; recruitment related to spawning stock size versus recruitment related to environmental conditions.

Consequences of managing for spawning stock size to future recruitment dependent upon the factor regulating recruitment

	Factor regulating recruitment	
Management approach	Spawning stock	Environment
Ignore spawning stock size	Risk of crashing the stock	Variable and unknown rate of recruitment
Manage for spawning stock size	Reduced risk of crashing the stock	Variable and unknown rate of recruitment

The prudent action under the conflicting hypotheses is to minimize the risk of crashing the stock. This would be achieved by assuming dependence of recruitment on spawning stock size, consistent with a *Precautionary Approach.*

There are two general classes of reference points:

1. Limits: set boundaries that define safe biological levels. Limits are often referred to as thresholds and are intended to minimize the risk of the stock falling below a minimum size (Mace, 1994; ICES, 1997).
2. Targets: are reference levels to aim for and are intended to meet management objectives such as achieving yields close to the maximum sustainable level (Mace, 1994).

Target reference points would be more conservative than limit points. Target mortality rates would be lower than the limit mortality rates whereas target spawner biomass reference points would be higher than limit spawner biomass levels. The management strategy would be designed to avoid exceeding the threshold and if the threshold is exceeded, then substantial reductions in mortality, including restrictions or prevention of the activity causing the mortality (for example fishing, turbine operation) would be considered (Rosenberg *et al.*, 1994).

Clear guidelines exist for the establishment and application of reference points (ICES, 1997):

1. A reference point is an estimated value derived through an agreed scientific procedure.
2. Both limit reference points and target reference points should be used.
3. Management strategies shall ensure that the risk of exceeding limit reference points is very low.

A large number of reference points (both mortality rates and biomass levels) and their associated data needs are summarized in ICES, 1997 (Table IX). The majority of reference points require information on several population parameters including age structure, growth, natural mortality, spawning stock size and recruitment size. The interpretation of some reference points versus a theoretical stock and recruitment relationship is shown in Figure 10.

The fishing mortality rate which generates maximum sustainable yield should be regarded as a minimum standard for limit reference points (ICES, 1997). To be consistent with the precautionary approach, limits should be defined in terms of mortality rates and spawning biomass levels (ICES, 1997).

There are advantages and disadvantages to the establishment and application of mortality rate limits and spawning biomass limits (Rosenberg *et al.*, 1994).

	Mortality limits	**Spawning biomass limits**
Advantages	Relate directly to the activity that can be controlled	Biomass is directly linked to recruitment
	Can be estimated from relatively limited data and information on life history characteristics	Provide a guide for management of stocks that are already depleted
	Can prevent stock depletion due to the long-term activity	Provides a seed stock for eventual recovery when adverse environmental conditions constrain abundance
Disadvantages	Do not provide protection for stocks which are already at low level	Difficult and extensive data to collect
	May require modification if environmental conditions and life history characteristics change	Risk of mis-estimation when a limited range of stock conditions is available
		May be mis-interpreted as the point at which the resource will collapse

5.1.2 Consequences of uncertainty

The greater the uncertainties, the greater the need to be precautionary (ICES, 1997). Increased uncertainty renders optimal harvesting strategies more conservative and optimal threshold increases (Lande, Sæter and Engen, 1997). F_{lim} and B_{lim} are reference points that should be avoided with high probability. There are uncertainties in the estimation of F_{lim} and B_{lim} as well as uncertainties in the assessments of the resource status relative to population abundance and exploitation. As a consequence of uncertainty, ICES (2000) defined precautionary reference points (F_{pa} and B_{pa}) to constrain exploitation ensuring a higher probability of not exceeding the limits. F_{pa} and B_{pa} are the main devices in the ICES framework for providing advice (ICES, 2000). For European eel, the degree of uncertainty is extreme and this should be reflected in the setting of reference points.

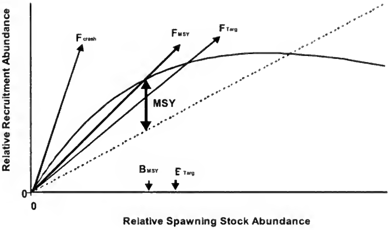

Figure 10. Position of some mortality rate and spawning biomass reference points relative to a theoretical stock recruitment relationship. The reference points are described in Table IX.

5.1.3 Proposed limit reference points in data-poor conditions

The majority of reference points require information on several population parameters including age structure, growth, natural mortality, spawning stock size and recruitment size. The limited knowledge and particular population dynamics of European eel are a major obstacle to the derivation of reference points. The wide distribution along the Atlantic and Mediterranean coasts of Europe and North Africa results in important differences in growth rates, age at maturity, and sex ratios. The mechanisms determining sex differentiation of animals are uncertain (growth rate, density, temperature, or a combination of factors). It is unclear how recruitment to freshwater occurs and whether there are regional stock and recruitment linkages. More important, there is little or no quantitative information on carrying capacity of habitat types for eels, or on what habitat variables determine carrying capacity. Natural mortality rates would vary with age and are likely to be high for the early life stages and decreasing with age and size.

Following the advice of ICES (1997), under data-poor conditions, a mortality rate which provides 30 percent of the virgin (F=0) SPR is a reasonable first estimate of F_{lim} until further information is gathered. Considering the many uncertainties in eel management and biology and the uniqueness the eel stock (supposedly single panmictic, spawning only once in their lifetime), a precautionary reference point must ultimately be more strict than the universal reasonable first estimate of F_{lim}. A preliminary estimate for F_{pa} could be 50 percent SPR.

Estimates of spawning stock and recruitment for the European eel are not available and are very unlikely to be feasible at all. Consequently, stock-wide management targets will have to be translated into derived targets for local management units (see Chapter 4). The

number of water bodies for which adequate information is available to warrant local management on the basis of fully documented assessments is extremely limited. In the absence of such data, ICES, (1997) suggested that biomass index series such as CPUE series, harvest rate models, or survey-based measures could be used to establish relative B_{lim} reference points. For example, the maximum survey index could be used as an indicator of virgin biomass and B_{lim} would be some value of that maximum level, such as 20 percent of max. The estimate of B_{pa} could be set at a value higher than B_{lim}, i.e. 50 percent of the maximum of the index series.

5.2 Preliminary reference points for European eel

5.2.1 Reference values

Mortality rate method

Preliminary mortality rate reference points could be established across the entire species range. Any reference points established should consider the following:

- Given the difficulties in estimating and forecasting stock size, fishing mortalities should remain below M (natural mortality) (Walters and Maguire, 1996),
- Uncertainty in estimated population size increased B_{lim} (Lande *et al.*, 1997),
- Ability to monitor compliance.

Overall loss to spawning stock depends upon the number of years eels are vulnerable to the fishery. Reference exploitation rates would vary with region – higher in the south than in the north.

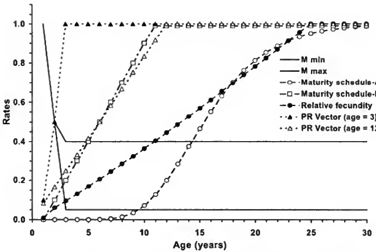

Figure 11. Input assumptions to the spawner to recruit modelling to estimate F_{lim} and F_{pa}. Maturity schedule refers to the proportion of the potential female yellow eels destined to metamorphose to silver eels. PR vector refers to the partial recruitment vector to the fishing gear. Maturation schedule A refers to a northern area stock and schedule B would be representative of a southern area stock.

Preliminary values of F_{lim} were derived from a theoretical recruit to spawner analysis (ICES, 2001) and provisionally determining F_{lim} at the F that generated 30 percent SPR. F_{pa} was estimated from the 50 percent SPR profile. These mortality reference points are estimated for eels aged one year and older. They could be calculated to include the recruiting stage but consideration for density-dependent regulation at the recruiting to yellow and silver eel stage would have to be considered.

The maturation schedule of eels is not well known, but female silver eels in northern areas are on average older than in the south. The same is true for male eels. Some representative maturation schedules were examined to see the effect of these on the SPR solutions (Figure 11). Simple partial recruitment vectors considered in the example calculations assumed full and constant recruitment at a given age (3 years or 12 years).

An example calculation estimating F_{lim} and F_{pa} for eel with variable maturation schedules probably typical of northern area and southern area stocks is shown in Figure 12. The percent SPR function is relatively insensitive to the natural mortality assumption (as seen by the width of the crescent profile) for the northern area assumptions but was more important for the southern area. F_{lim} to F_{pa} range was narrow (between $F = 0.06$ and 0.12) for the northern area stock and wider (between $F = 0.11$ and 0.32) for the southern area stock (Figure 12). The maturation schedule is particularly important in the estimation of F_{lim} and F_{pa} as this determines the number of years the animal is exposed to the fishery.

The reference points are also sensitive to the partial recruitment vector assumption. The partial recruitment profile would respond to management actions such as size limits on retained eels, mesh size limits, area restrictions, and seasons. In the second example, the effect of different partial recruitment vectors (fully recruited at age 3 years versus fully recruited at age 12 years) (Figure 11) but for a fixed maturation schedule (northern profile) is described. The F_{lim} and F_{pa} points increase as the age of full recruitment to the fishery increases.

Biomass method

No precedent exists for the setting of biomass SSB limits for eel. However, the method is an accepted method of assessing marine fish stocks, along with known or estimated SSB to recruitment relationships.

Most European eel producing areas are extremely data-poor, with insufficient data for stock assessments based on standard methods. Thus, other means of formulating reference points are required, at least *ad interim*, until data sufficient for the practical implementation of traditional stock assessment methods become available.

A provisional limit reference point is therefore proposed based on the contribution of individual catchments to the spawning stock relative to the notional biomass of an unfished stock in an environment with no negative human impacts, such as habitat loss or degradation and mortality to spawning migrants during downstream passage through power generation turbines.

The carrying capacities of freshwater habitats for European eel have been reviewed by Moriarty and Dekker (1997) who assumed an average of 10 kg/ha. This value could be used as a benchmark value, with a proportion assigned as an initial reference level for the minimum silver eel escapement for each catchment or chosen geographical area. However, the best habitats can produce 40 kg/ha or more and there is a south-to-north decrease in potential silver eel production. Thus, the potential minimum output from rivers in the Mediterranean region decreases from about 40 kg/ha, through 20 kg/ha along the continental Atlantic coasts, to 10 kg/ha in the southern North Sea and British Isles, and to about 5 kg/ha

in the Baltic and Swedish/Norwegian rivers. Escapement conservation limits would be set as a proportion (e.g., 30 percent) of these regional potential production figures rather than of the European average value. The primary management objective would therefore be to ensure a high probability of maintaining the spawning escapement above these limits.

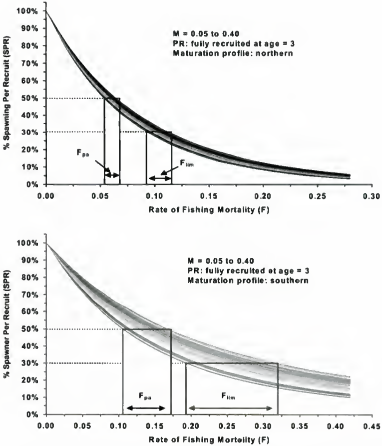

Figure 12. Estimated percent SPR relative to F for the eel stock of the northern area (upper panel) and a southern area (lower panel) for varying assumptions of M. The estimated percent SPR is eggs per R (adjusted for fecundity at length).

Dekker (1999) estimated a Europe-wide spawning escapement limit reference value of 23 to 33 percent of potential unexploited spawner biomass. Setting the conservation limit at

30 percent of potential unexploited spawner biomass would ensure that spawning escapement is sustained. The threshold level is expressed as a percentage of the level that would occur if there was no fishing and is referred to as the 'threshold replacement percent SPR'. For species for which very limited stock-recruitment data are available it may be appropriate to set a threshold replacement percent SPR of 30 percent. The conservation limit proposed is more risk averse than a 30 percent SRP. This is because the conservation limit is based on the ratio of two biomass (or stock) levels, while the percent SPR is based on the ratio of two spawner-per-recruit levels. If recruitment is not substantially reduced when the spawning stock level is reduced to 30 percent of the unexploited level, then an escapement of 30 percent will be equivalent to a percent SPR of 30 percent. If, however, the recruitment is reduced, then an escapement of 30 percent will translate into a percent SPR greater than 30 percent.

Length-frequency distribution reference point

Length-frequency distributions are perhaps the most common of eel data sets. They offer the possibility of a simple reference point related to the numerical proportion of potential emigrants. The fishing of yellow eels tends to crop the larger individuals, resulting in size distributions skewed towards smaller eels. This suggests the possibility of estimating fishing mortality, biomass reduction, SPR reduction, or other effects of fishing from length-frequency analysis. In principle, an arbitrary limit reference point can be proposed, e.g., that 50 percent of eels should exceed 50 cm and thus be potential spawners. This approach has the advantage that it would result in collection of data that could enable year-to-year refinement of life table models and could ultimately lead to mathematically-based escapement estimates. However, no length-frequency distribution reference point can be recommended at this time because of insufficient testing of the method.

5.2.2 Limitations of methods

Limitations of F_{lim} method

The estimates described above are based on equilibrium conditions, i.e. no change in characteristics with abundance. Adding stock and recruitment to the model has an effect on yield calculations, i.e., yield declines with increasing spawning stock size (Hilborn and Walters, 1992). Defining only F_{lim} reference levels can be dangerous because an F-based definition appropriate over a middle range of biomass levels may not be appropriate at the extremes of biomass. Also, the definitions set to prevent long-term decline of the stock do not increase the protection to the resource when it is in poor condition (Rosenberg *et al.*, 1994).

Limitations of the biomass method

There are undeniable problems in setting a reference point based on a proportion of the SSB expected for any given system in the absence of fisheries and other deleterious impacts. The central problem is the definition of pristine habitat and estimation of biomass under unfished conditions. Most eel producing freshwater systems have suffered habitat reduction or degradation in habitat quality. Thus, the starting point should be based on <u>full utilization of currently available</u> eel habitat. The provision of access to additional eel habitat upstream of barriers is a practicable option for increasing SSB outputs in many systems. Most eel stocks of reasonable abundance are also fished and the stock structure and biomass available under unfished conditions may be difficult to quantify.

The second problem encountered is in measuring output. This imposes a requirement for field study monitoring. Some freshwater fish stock monitoring of resident fish stock biomass, including for eel, will be required under the EU Water Framework Directive. This, in conjunction with length-frequency data or age profiles, and known proportions of emigrants based on the results of some current field programs (in France) could form the basis

of estimating individual system escapement of SSB in the near future. Where large-scale fisheries for silver eel still exist, mark-recapture programs offer another means of assessing SSB escapement.

Generation of the data series to allow adoption of this method to eel stocks will take many years. The co-ordinated development of an annual combined European SSB time series and continuation of existing recruitment time series would eventually allow the examination of SSB to recruitment relationships.

Limitations of the length-frequency distribution method

Problems with the length-frequency distribution method include the known naturally downward skewed size distributions from electrofishing data in shallow streams and, in some cases, in lower reaches of rivers. Length-frequency distributions can also be temporarily skewed downward by recruitment of a strong year class. Natural variations in recruitment may also mask biomass changes due to fishing pressure. The measured populations are also assumed to have no immigration or emigration during the continental juvenile phase. Different applications of the method may require data such as a time series of measured abundance, the length frequency distribution and age of the stock, and data on the unexploited stock.

5.2.3 Application of limit reference values

Application of F_{lim} and F_{pa}

The estimation of F_{lim} and F_{pa} levels applicable to a stock are dependent upon information on the age and size composition, maturation schedule, and characteristics of the fishery itself including size selectivity and availability of life stages to the gear.

When the mortality factors on the stock are managed such that F equal to or less than F_{pa}, there should be a low probability that the realized mortality is not sustainable (ICES, 1997). In the absence of B_{lim} and B_{pa} reference points, other measures of stock status would be used to assess compliance with the limit and PA points. These indicators would include size composition of the catch relative to unexploited areas, relative abundance of yellow and silver eels (when these are available for capture), condition factors, etc.

Application of the biomass method

The biomass method has not yet been applied to a specific eel stock. Application of the method depends on developing an acceptable definition of pre-exploitation available habitat and of the biomass produced under those conditions as well as an estimate of current biomass under existing fishing conditions. The target and limit biomass levels appropriate to eel stocks require further development. For the meantime, a target biomass level is proposed of 30 percent of the unexploited biomass level.

Application of the length frequency distribution method

A simulation model for New Zealand eels by Francis and Jellyman (1999) found that only large (>40 percent) changes in biomass could be detected by shifts in mean size. A stochastic life table model (see Chapter 7.2) may be used to track cohort strength and demographic factors between glass eel arrival and egg deposition. This model differs from the Francis and Jellyman (1999) approach in that the slope of the right-hand limb of the length-frequency curve was used to infer stock parameters. Simulation results showed that recruitment variation strongly affected the frequencies of smaller eels but had little effect on frequencies of larger size classes.

The model requires a length-at-age table, an equation relating length and weight, and the length frequencies of unexploited and exploited populations. It can be used to estimate fishing mortality and summed natural mortality/emigration. The plot of the relation between fishing mortality and percent reduction in spawn output per recruit (SPR) allows determination of the fishing mortality that corresponds to a given SPR-based conservation reference point. In this model, SPR is defined as the reduction of egg deposition due to fishing. Where fecundity-weight relations are unknown, SPR reduction could be modelled in terms of biomass of emigrating females.

This model permits estimation of key demographic parameters and evaluation of compliance with conservation reference points using relatively modest data requirements (lengths, weights and ages). The model examined in Chapter 7.2 found the slope of the descending right-hand limb of the length frequency distribution was about 3 times steeper for a relatively heavily exploited population than for an unexploited population. The slopes of the length frequency distributions of presently exploited stocks may be indicators of exploitation status.

5.3 Conclusions

Essentially, there are two possible approaches by which spawner escapement targets might be set for European eel in specific river systems:

1. mortality limits
2. spawning biomass limits

In reality, most systems have insufficient data on which to set escapement targets based on either of these two. Only the few data rich systems can allow ready implementation of any limit reference value. In the long term, mathematical models may be developed for the total stock based on individual monitoring of its components, and the methods chosen now to set limits should encourage the collection of the necessary data and to derive proximate criteria for data-poor environments.

Provisional targets needed and developed now will necessarily be based on a very high degree of uncertainty in the available data. Provisional targets must be workable in extremely data-poor systems and will have to permit a high degree of local management flexibility.

Thus, the only possible basis for immediately applicable escapement targets may be biomass limits set as a percentage of the theoretical pristine silver eel outputs for major river systems or groups of river systems. The initial target proposed is 50 percent of this theoretical production level. The theoretical pristine production figure must eventually be set on a local or regional basis. Additionally, limits based on length composition of the catch can be applied, but these limits are currently still highly arbitrary.

Implementation of stock monitoring to define pristine condition spawner outputs requires fish population surveys for length/age frequency distribution, sex ratios, and abundance/density estimates. These data, in conjunction with present and continuing time series of eel recruitment and production will, if systematically gathered across the range of the eel, lead to development of better models of eel population dynamics and stock-recruitment relationships.

It is important to note that EU member states will soon be required, under the Water Framework Directive currently being implemented, to gather much of this data and to use it to assess habitat condition relative to a notional pristine or reference eondition.

6 PROPOSED MANAGEMENT ACTIONS

6.1 Introduction

The FAO Technical Guidelines on the Precautionary Approach to Capture Fisheries (FAO, 1996) call, among other things, for *'prior identification of undesirable outcomes and of measures that will avoid or correct them'*. It is thus desirable that managers determine how they will react to a problem before it occurs, and that management strategies should define the actions that will be taken if, for example, stocks approach or fail to meet limit reference points. Without such a predetermined decision structure a tendency for social and economic justifications may be used to water down or delay management actions. This may present particular problems for the management of the European eel because stocks are already in a depleted state.

The application of a precautionary approach requires that *'any fishing activities must have prior management authorization and be subject to periodic review'*. A concern in this regard is that managers in many parts of Europe may currently be poorly placed to regulate and monitor eel fishing activities, due to only limited, if any, mechanisms for controlling effort or catches in eel fisheries. Providing an appropriate legislative framework, whereby appropriate controls can be introduced, within the context of a wider European strategy, should therefore be a high priority for managers.

The management and conservation options available to managers in a number of European countries have previously been summarized by Moriarty and Dekker (1997). These highlighted the marked regional variation in approaches, reflecting the widely differing traditions relating to both, eel fishing and consumption, in these countries. More recently, in light of the continuing decline in recruitment, some countries have introduced additional measures. In particular, measures have been proposed/introduced to restrict glass eel fishing by tighter controls on fishing by unlicensed (non-professional) fishermen (France) or by the prevention of any extension of glass eel fishing into new fishing areas (England and Wales).

In England and Wales a national eel management strategy has been introduced (Environment Agency, 2001), which sets out a framework for management of national eel fisheries and populations in the light of the need for a precautionary approach. Key objectives of the strategy are:

- improved stock and fishery sustainability (recognising the need to work towards appropriate conservation limits);
- an improved legislative and regulatory framework; and
- increased knowledge and awareness (including specific recommendations for monitoring).

It should be noted that aspects of this plan take a long-term view and that full implementation will be dependent on available resources. There have been no similar initiatives in other European countries, although national 'reviews' have been completed in some cases (e.g. Sweden, the Netherlands, Brittany/France). However, there is widespread recognition of the need for a co-ordinated, over-arching European management strategy for the eel that will apply to all life stages and fisheries across the range of the species.

6.2 Management actions that may lead to the required escapement

With reliable data on catches, effort and the status of stocks it would be possible to consider long-term management, define well-derived reference points for fishing mortality and spawning stock biomass and co-ordinate management efforts across the range of the

European eel. However, the current data-poor situation requires a pragmatic approach before such facts and figures are available.

Where stocks are depleted, application of the precautionary approach requires that recovery plans are set up to restore stocks quickly, i.e. commonly 1-2 generations, for eel equal to a time span of 15 to 20 years. This is probably an appropriate time period to have in mind as a basis for management.

A range of factors, other than just fisheries, are likely to be involved in the decline of the European eel stock, and action is likely to be required in many areas. For example, to improve or increase access to freshwater habitats (e.g. Moriarty and Dekker, 1997; Knights and White, 1998; ICES, 2001), or restrict the impact of hydro-power installations or other anthropogenic impacts (ICES, 2001). Thus the application of a precautionary approach to the management of eels should not only affect the regulation of fisheries; it should also relate to non-fisheries factors, such as the management of freshwater, estuarine and coastal habitat. It may also require attention to other activities such as aquaculture insofar as this can affect, for example, market forces, transfer of recruits and the possible introduction of diseases and parasites.

Management options (discussed in more detail below) include measures to limit exploitation by fisheries, protect and improve the productive capability of eel habitat, and enhance production through expansion of accessible habitat and the stocking of under-utilized or inaccessible habitat.

6.2.1 Measures to limit exploitation by fisheries

Measures to limit exploitation by fisheries will commonly be site/area and circumstance specific and will generally function by regulating the length of time that individual eels are potentially vulnerable to fisheries. Consideration may also need to be given to the potential volatility of eel market demands and hence possible short-term but large fluctuations in fishing pressure.

Prohibition of fishing

Prohibition of fishing can be life-stage specific or area specific. For example, commercial glass eel fishing is banned in countries where supplies are low (Sweden, Denmark, Germany, N. Ireland, Ireland, the Netherlands, Belgium). Within England and Wales precautionary measures have been drawn up to introduce a byelaw to limit glass eel fishing to the principal existing fishing zones, thus preventing further expansion of the fishery. It is prudent to prohibit extension of existing fisheries and introduction of new fisheries in England and Wales as well as elsewhere.

Total allowable catches/quotas

Ideally, application of total allowable catch/quota restrictions requires knowledge of abundance and identification of escapement targets. Quotas put an upper limit on the total catch; however, with the diverse nature of eel fisheries, it is difficult to envisage how an individual quota on a panmictic stock would be shared and subsequently managed/enforced in the scattered inland fisheries in Europe. Required data are not currently available for different life-stages of eel and therefore TAC approaches are probably not workable.

Gear controls

Controls on, for example, number, size, mesh-size, usage and location of gear are already enforced in several eel fisheries to control fishing mortality. For example, in the Severn, fishing for glass eel from a moving boat is not allowed and only hand-held dip nets of

a set size can be used. In some silver eel fisheries (e.g. Lough Neagh) gaps have to be left to allow some escapement. Where they do not exist, gear controls should be introduced and in other areas strengthened.

Landing size limits

Minimum size restrictions could help to reduce excessive exploitation of yellow and pre-spawner eel; such measures are already in place in some countries (e.g. the Netherlands, Ireland, Denmark) and have recently been strengthened in Sweden. Minimum mesh size limits for fyke and other nets have been set in many areas. Limits on maximum size would promote escapement of larger (female) pre-spawners, but could also result in increases in fishing effort aiming at depletion of the stock of smaller sizes.

Closed seasons

These are currently in operation in some countries, but are commonly based on traditional or practicable fishing season (e.g. Ireland) or are primarily related to requirements to allow unhindered migration of salmonids (e.g. Denmark and N. Ireland). The effectiveness of fishing time controls is affected by temporal variations in eel activity and migrations, often as a result of changing environmental parameters. Only banning of fishing over relatively long time periods would be fully effective, e.g. if extending beyond the duration of local glass eel immigration or silver eel emigration runs. The timing of closed seasons must be related to local characteristics of eel and fisheries, and has to primarily consider closure during periods of vulnerability. There are no seasons in which all eel fisheries in Europe are in operation.

Closed areas

These could be locally effective, e.g. in preventing extension of fisheries (particularly for glass eel/elvers) into new areas or for protection of vulnerable glass eel/elver or silver eel runs. Alternatively, closed areas could be used to designate 'reserve' or 'refuge' areas where no exploitation would be permitted. Such an approach is currently used in the management of eel stocks in New Zealand and could be applied to watersheds in parts of Europe where unexploited eel populations are known to exist where the simplicity of closed areas is preferred over other regulations, more difficult to control.

Licensing of fishermen and dealers

Licensing specific to eel fishermen and their gear and dealer licensing could help provide, via catch returns and market statistics, improved information for monitoring catches and compliance with targets. The quality of such information is currently often poor, but licensing of fishermen and gear, in conjunction with adequate enforcement of regulations, offers opportunities for controlling and monitoring fishing effort and, ultimately, fishing mortality. In England and Wales it is planned to introduce a revised system of licensing and compulsory catch returns for fishermen in the near future.

6.2.2 Measures regarding eel habitat

Measures should be taken to insure and promote the access of eels to all reaches within catchments. The higher natural quality of freshwater catchments will promote healthy eel populations. Proper enforcement of the EU Water Framework Directive and guaranteeing full accessibility of eels to freshwater habitats should be a management priority

Eel management needs to be considered at the minimum level of the river basin scale, from the estuaries to the sources and from the river basin (including land biogeochemical cycles of contaminants) to the estuaries. In particular, it is necessary to improve the measures

and technology to protect, manage, enhance and restore habitats; to ease migration and movements of eels, upstream (accessibility) and downstream (escapement of silver eels and contribution to spawning stock).

These proposed actions meet the requirements of the EU Water Framework Directive, which states that '...EU countries should prevent further deterioration of their waters as well as protect, restore and enhance and restore them in order to achieve good or high ecological status in all their water bodies. To achieve this goal, countries must begin developing river basin management and monitoring programmes...'.

Insure habitat accessibility

This should be achieved by increasing the number of fish passages in existing dams and insuring that new dams are equipped with passages. The highest priority should be given to those in the lower part of rivers that block or hamper the early ascent of glass eels. The effectiveness of old and new fish passages in allowing the migration of eels should be measured and improved where necessary. Measures to insure and promote the maintenance of eel passages should also be enacted.

Reduce habitat loss

Measures to protect existing wetland habitats should be taken, since these areas sustain considerable eel stocks. Efforts directed towards the restoration of wetland habitats and degraded river sections will augment existing eel habitats and ultimately result in increased escapement figures.

Insure habitat quality

Measures should be taken to restore habitat quality, chemically and ecologically. This involves collaboration with ongoing restoration efforts as recommended by the EU. Improvements of eel habitat quality is assumed to increase the breeding potential of spawners.

Insure downstream migration

Up to date, no measures have been taken to reduce mortality of downstream migrating silver eels through hydroelectric turbines and dam bypass systems. Efforts to insure upstream migrations of glass eels or elvers and restocking efforts can be futile if downstream migration of eels is not insured. Mortality of downstream migration of silver eels across dams should be minimized by the construction of properly designed downstream passes for silver eels. These measures should be taken into account when building new dams. Similar mortality reducing measures should be also be applied to existing dams. Both, technical measures and management procedures can be utilized to insure minimal mortality levels. Such management plans should be designed for complete river systems in order to restore downstream migration from upper reaches to the ocean.

7 SCIENTIFIC BASIS FOR ADVICE

7.1 Introduction

In sharp contrast to assessment information collected for species populations with a relatively narrow range, the widespread but fragmented spatial distribution of European eel is such that truly representative monitoring may not be achievable. Many life history characteristics vary throughout the distribution range. The scale of impacts to eel life history varies widely from localized to oceanic levels. International research programmes on eel require an international co-ordination framework. Coherent research plans focusing on stock-wide management have been prepared (EIFAC, 1993; Moriarty and Dekker, 1997;

ICES, 2000), but actual research programmes have been influenced only marginally. A co-ordinated, international effort to collect relevant data would allow better management advice to be given than is currently possible.

Therefore, it is strongly recommended that an international commission be formed to organize monitoring and research. The commission would serve as a clearing house for regular exchange of information regarding landings and resource status, and it would provide insight on research needs.

So far, internationally co-ordinated studies, such as the EU 1993 Concerted Action on Management of the Eel and the running EU 1998 Concerted Action on Establishment of a Recruitment Monitoring System, have depended entirely on the initiative of concerned scientists, have relied amongst others on national research budgets and have not covered execution of basic monitoring and continuation of existing data series.

All current monitoring is based on national management interest only. Several of the long lasting series have come under pressure of budget cuts, because of the low state of the local eel fisheries and the impossibility of addressing the stock decline at the local level properly. The responsibility for the management of the stock far exceeds the competence of the local authorities. The continent-wide monitoring programmes needed for stock-wide management require continued concern of local and trans-national managers. However, in practice it is rather difficult to attract the attention of managers to the monitoring of a stock which is currently in severe distress and therefore of little economical importance.

In recent years, monitoring of recruitment at Imsa (Norway), Vidaa (Denmark), Ems (Germany), IJser (Belgium) and Nalon (Spain) have (effectively) been discontinued, and at Tiber (Rome) and Den Oever (Netherlands) have come under financially motivated pressure. Lack of progress in the development of an international management plan for the European eel has been cited as an argument. Landing data provided by several major eel fishing countries (Italy, the Netherlands, Denmark) to FAO Fishery Information, Data and Statistics Unit have become unreliable, because of mixing of fisheries and aquaculture production. Consequently, the situation of inadequate or insufficient documentation on the status of the stock is rapidly deteriorating.

Therefore, it is strongly recommended that the development of a stock recovery plan is taken up as a matter of urgency and meanwhile current monitoring efforts are sustained at least at present/recent levels.

7.2 Development of harvest rate models

To be both, biologically realistic and widely usable, an eel population model must embrace the eel's peculiar demographic features, including high variability in growth rate and its consequences, and also be capable of implementation with limited data. This section presents a stochastic life table model in which natural mortality and maturity schedule depend on size, and size at age varies according to a randomized growth function. The model is suitable for use where eels are exploited at the yellow stage, and growth and mortality are not controlled by density-dependent factors. Data requirements are length at age, a length-weight relationship, and length-frequency distributions for exploited and unexploited populations. The model estimates fishing mortality and summed natural mortality/emigration rate, and evaluates compliance with conservation reference points based on spawning per recruit (SPR) reduction as a function of fishing mortality.

7.2.1 Model structure and inputs

Data from American eel populations on Prince Edward Island (PEI), in the southern Gulf of St. Lawrence, Canada, are used as inputs to the model. Eels are fyke netted in PEI tidal estuaries and adjoining bays between mid-August and mid-October. The minimum legal size is 50.8 cm. In the estuaries of the Pinette River system, sampled eels were smaller in 1973, when the area was exploited, than in 2000, when no fishery existed (Figure 13). Similarly, eels sampled in exploited estuaries in 1997-2000 were smaller than Pinette eels sampled in 2000 (Figure 14). Length frequencies from exploited estuaries declined with a slope of -1.38 between the modal length and the point where the percent length frequency fell below 2.5 percent. For the unexploited (Pinette) estuaries, the slope was -0.43.

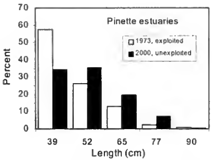

Figure 13. Length frequency distributions of American eels sampled in the estuaries of the Pinette River system, Prince Edward Island, Canada in 1973 when the site was commercially exploited and in 2000 when it was not.

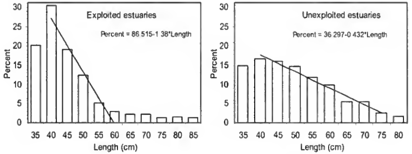

Figure 14. Frequency distributions of American eel lengths in exploited (n=2284) and unexploited (N=531) estuaries of Prince Edward Island. Percents are based on eels > 35 cm long. Regression lines are for percent frequencies in the range between the modal length and the point where the frequency falls below 2.5 percent.

A von Bertalanffy curve was fitted to length at age of 130 eels sampled from marine and freshwater habitats by using Microsoft Excel Solver to minimize squared residuals (Figure 15). Weight (W, in g) of PEI eels is related to length (L, in cm) by the equation $W = 0.000535 \times L^{3.3089}$ based on measurements of 2668 eels.

The life table model tracks the major demographic processes of eel cohorts between arrival at the coast as glass eel and egg deposition in the Sargasso Sea. The model assumes that glass eels arrive on 1 June, and that all glass eels are destined to become female.

During their continental residency modelled eels grow in length according to the von Bertalanffy equation for PEI data (Figure 15). Variability in growth rate is achieved by varying the L coefficient of the von Bertalanffy equation according to a normal distribution. The coefficient of variation of the von Bertalanffy L term was adjusted until modelled length outputs for age 3 eels had the same coefficient of variation as lengths at age 3 in aged PEI samples (Figure 15). Figure 15 also illustrates the scatter of lengths at age produced by the randomization procedure. Weight is calculated from length according to the PEI length-weight equation.

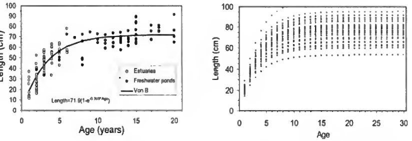

Figure 15. Length at age of American eels. Left panel: data for eels from Prince Edward Island, with a von Bertalanffy curve fitted. Right panel: length at age simulated by the life table model. Lengths are generated by the von Bertalanffy equation with the L term varied according to a normal distribution which produces the same coefficient of variation (0.15) of lengths for age three in the simulated population as was found in the real data.

Natural mortality (M) in fish depends closely on weight, and can be modelled through allometric equations of the type $M = aW^b$. M is modelled with Lorenzen's (1996) equation $M = 3.00 \cdot W^{-0.288}$. The exponent b of M-W equations is assumed to be relatively uniform (McGurk, 1996), but the coefficient a may vary. Hence in the life table model the exponent b (-0.288) is held constant, but the term a (3.00) is multiplied by adjustment factors. This method is used to calculate M for all stages between glass eel arrival and female spawning.

The run of juvenile eels to the Petite rivière de la Trinité in the north-western Gulf of St. Lawrence was estimated in the mid 1980s and emigration of silver eels was estimated in 1999 (ICES, 2001, see also Fournier and Caron, 2001). About 2 percent of the estimated juvenile run survived to leave as silver eels. Application of the unadjusted Lorenzen (1996) equation produced a cumulative survival of only 0.002 percent. When the natural mortality

coefficient was multiplied by 0.164, the cumulative survival became 2 percent. An adjustment factor of 0.164 was therefore adopted as a starting point for mortality analysis.

Modelled eels are subject to fishing mortality after attaining 50.8 cm, the minimum retention size on PEI. The model assumes that a user-specified proportion of eels emigrate to the spawning ground after a threshold size is reached. This threshold was set at 50 cm, based on the appearance of silver coloration in eels this size and larger on PEI. Emigrating eels depart on 2 October, and spawn on 7 February. Fecundity is calculated from weight by Barbin and McCleave's (1997) formula ($F = 14608 \times W^{0.9153}$).

The life table model was prepared in two versions. Version I tracks a single cohort of 1 million glass eels through its life cycle. Model output is the aggregate sum of 1,000 trials (except in SPR analysis when 10000 runs were used). Version II tracks the fate of cohorts that arrive in 20 successive years. Initial cohort strength (mean 1 million) is randomly varied according to a normal distribution whose coefficient of variation (0.50) matches that of the elver run in East River Sheet Harbour, Nova Scotia (N=10) (ICES, 2001). Each year is assigned a randomly selected cohort population, which it retains in each of 1000 runs. The model compiles demographic data on eels that are alive in year 20, derived from glass eel cohorts that arrived in each of the 20 previous years.

7.2.2 Model output

Natural mortalities and emigration rates were adjusted in Version I of the life table model to seek combinations that yield length frequencies whose right-hand limbs have slopes that resemble those of real data. When the natural mortality adjustment factor was set at 0.164 (as estimated for the Petite Trinité, see above), an annual emigration rate of 18 percent above 50 cm yielded a length frequency whose right-hand slope resembled that of the unexploited population (Figure 16). When the natural mortality adjustment factor was 0.5, an emigration rate of 13 percent produced a distribution whose right-hand slope resembled that of the distribution for unexploited eels. When fishery mortalities were introduced, the slope of the right-hand limb of the frequency distributions steepened. Under both adjustment factor assumptions (0.164 and 0.5), the slope most closely resembled those of exploited populations when F was set at 0.60 (Figure 17). This suggests that eel fishing mortality in the PEI eel fishery is approximately 0.6.

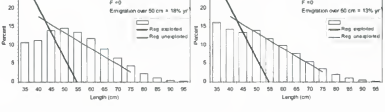

Figure 16. Length frequencies of resident American eels calculated by the life table model for unexploited populations. Adjustment factors for mortality equations and emigration rates have been adjusted so that the slope of the declining limb of the length frequency distribution matches regression lines for length frequencies from an unexploited population.

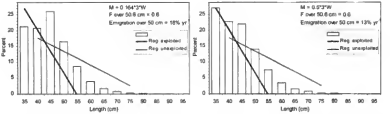

Figure 17. Length frequencies of resident American eels calculated by the life table model for exploited populations. Adjustment factors for mortality equations and emigration rates are as in Figure 16, and fishing mortality has been adjusted so that the slope of the declining limb of the length frequency distribution matches regression lines for length frequencies from measured exploited populations.

Version I of the life table model is based on single cohorts, but actual eel populations at any given time consist of multiple cohorts, derived from initial populations that may have varied inter-annually. If length structure is largely determined by cohort size variation rather than by mortality and emigration rates, then the estimation of mortality and emigration rates from the length frequency outputs of life table models would be invalidated. To examine the effects of inter-year variability in recruiting cohort size on population length structure, Version II was run 10 times with a mortality adjustment factor of 0.164 and an emigration rate of 18 percent (Figure 18). Frequencies of lengths under 50 cm varied substantially among runs, but at greater lengths, length distribution showed relatively little inter-run variation. Trials with an adjustment factor of 0.5 and an emigration rate of 13 percent also showed little inter-run variation in distribution of eels above 50 cm. This suggests that comparisons between simulated and measured length frequencies can be used to estimate mortality and emigration rates, provided that comparisons are based on size classes above 50 cm.

Effects of fishing mortality on spawn output was modelled in Version I by calculating egg deposition as a percent of egg deposition in an unexploited population. Total egg deposition was modelled rather than female escapement. Total egg deposition reflects contribution to the next generation better than numbers of escaping females because fishery regimes affect size distribution of female escapees, and sizes influence mortality rate during transit and fecundity.

Two scenarios were modelled. First, the natural mortality adjustment factor was 0.164 and the emigration rate over 50 cm was 18 percent, and second, the adjustment factor was 0.5 and the emigration rate was 13 percent. In both cases, the percent of maximum egg deposition declines at first sharply with increasing F, and then more gradually at higher F's (Figure 19). At a given F, percent of maximum egg deposition was less when the adjustment factor was 0.5 than when the adjustment factor was 0.164.

A 50 percent reduction in egg deposition (F_{pa}) was reached when F was 0.16 and 0.2 for mortality adjustment factors of 0.164 and 0.5, respectively. A 70 percent reduction in egg deposition (F_{lim}) was reached when F was 0.34 at adjustment factor 0.164 and 0.42 at adjustment factor 0.5. At the estimated F for the PEI eel fishery (0.6), reductions in egg

deposition were 83 percent for the 0.164 adjustment factor and 79 percent for the 0.5 adjustment factor.

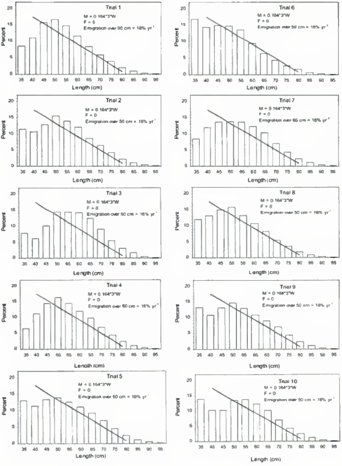

Figure 18. Length frequencies of American eel populations simulated by the life table model. Recruiting populations vary annually, with a coefficient of variation of 0.5. The straight lines are from the regression equations for length frequencies of unexploited eel populations.

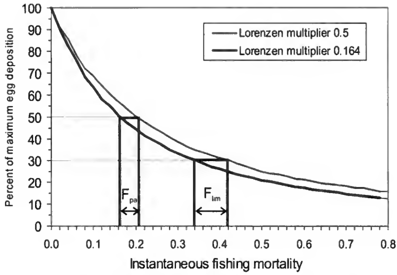

Figure 19. Relation between fishing mortality and egg deposition in American eels of Prince Edward Island as a percent of egg deposition in the absence of exploitation. Natural mortality is given by Lorenzen's (1996) weight-based formula, adjusted by the multipliers given in the legend

7.3 Migration

7.3.1 Fish passes for upstream migration of recruits

The efficiency of fish passages on upstream migration has been shown by a number of studies. For example, in the Vilaine River, a dam was built in the early 1970s and consequently, the eel stock was very much depleted in the 1980s. An eel pass was built in 1995 enabling a tenfold increase of the eel densities and an extension of the distribution area (Briand and Fatin, 1999; Briand *et al.*, 2000a). In some small coastal river systems, eel passes enable to maintain eel stocks and distribution at their carrying capacity.

Eel ladders enable passage of important quantities of eels right up to the upstream reaches of large rivers (Legault, 1994). However their efficiency still needs to be improved as studies showed that only 30 percent of eels used the ladders, the remainder staying downstream of the obstruction and thus being subjected to increased mortality rates (Briand *et al.*, in press, a; Briand *et al.*, in press, b).

7.3.2 Downstream migration of silver eel

The technology concerning downstream migration, and mitigation of mortality through turbines and hydraulic by pass systems is very poorly known (Legault *et al.*, in press). The technology to reduce mortality through turbines is not cost effective as the only efficient solutions which were proposed up to now were to reduce significantly the water flow through turbines therefore resulting in drop of electricity generation. Therefore models were

developed to predict periods of migration peaks during which electricity generation would be reduced which would favour escapement over the dams (Feunteun *et al.*, 2000a). Another solution proposed by Electricité de France was to develop a silver eel fishery in heavily developed rivers, as Rhône or Rhine, in order to translocate eels downstream. On a 14 m dam built for water supply a by pass system was installed to insure minimum legal water discharge. This system which originally created about 100 percent mortality was modified and silver eel mortality was reduced to about 10 percent and enabled the passage of about 10 percent of the migration runs (Legault *et al.*, in press).

7.4 Habitat improvement

Considering eels are highly ubiquitous species, they colonize practically every kind of water body available and accessible over the distribution range. They especially invade and establish permanently in wetland habitats as river flood plains, coastal marshes in marine areas or lakes and lagoons. These areas are known for their high productivity and related trophic value (consistent food supply). Therefore, they are able to host dense eel stocks of 50-300 g/m^2 (Feunteun *et al.*, 1999). Experiments show that restored water bodies and wetlands, provided they are correctly connected to migration routes, are rapidly colonized by dense populations of eels. For example, in the Brière Marshes, 300 ha of water bodies were restored to mitigate effects of land abandonment. These habitats were rapidly colonized by a dense eel population (Eybert *et al.*, 1999). In coastal marches of western France which were obstructed by silt because of shift in management practices (Feunteun *et al.*, 1992) a project was conducted to restore pristine habitat conditions over 350 ha. The eel population rapidly approached prior reference levels of about 50 kg/ha (Baisez, Rigaud and Feunteun, 2000).

8 FURTHER DEVELOPMENT OF ADVICE ON EEL

8.1 Interaction between management and research

In comparison with other species, the management of eel stocks and fisheries is rather complicated. Several facets of the basic biology of the species are unknown, and biological characteristics of the eel vary from region to region and from habitat type to habitat type. The stock and fisheries are distributed over most of Europe, northern Africa and a minor part of Asia. In contrast, many typical eel fisheries operate in small water bodies, fished by a few fishermen at a time, from which hardly any information is derived. Consequently, establishment of a management system for the eel cannot proceed along the same lines as for other, more typical marine or freshwater species. It has been recommended that a stock recovery plan should be compiled. Completion of such a plan will necessarily entail additional research and monitoring, to clarify uncertainties and to investigate unknowns.

Although the advice given in this report is based on prolonged discussions on required and feasible management regimes, it is recognized that further development of the advice, and furthering our scientific knowledge, cannot proceed without close co-operation between managers and scientists. The periods between the first observation of the eel stock collapse (1985), the first management advice (1996), the compilation of comprehensive research plans (1997), the urgent recommendation to compile a stock recovery plan (1998) and the ultimate implementation of stock-wide management measures influencing the eel stock and fisheries (when?) do not encourage an optimistic view. Improvement of the advice depends crucially on agreement to, and implementation of, an international management process with appropriate feedback to scientific advisory bodies.

8.2 Facilitation of provisional management measures

It is recommended that provisional limit reference levels are set in respect of exploitation of the European eel. Additionally, it is recommended that the effect of habitat loss (upstream or downstream migration barriers as well as physical loss of habitat) on the production of spawners is given due consideration. Noting that the continental eel stock is fragmented over myriads of water bodies, in thousands of jurisdictional entities, implementation of provisional targets would be greatly facilitated by the development of practical guidelines for managers. This might include advice on: implementing a management regime; options for monitoring and fisheries management; building of fish passes and downstream migration facilities; habitat restoration; etc. This should also include recommendations for the development of proximate criteria, for the few data rich situations as well as for the most common data-poor conditions. It is recognized that the implementation of limit reference levels and controls on exploitation will probably have socio-economic implications, especially since eel fisheries play a crucial role in coastal rural communities. It is therefore recommended that socio-economic effects are also considered further.

At the international level, management targets will have to be defined and refined. The current advice sets limits relative to the unexploited state, although this is not clearly quantified. Investigations of unexploited systems as well as analytical studies of exploitation by fisheries might fill this gap. Subsequently, procedures will have to be developed for post-evaluation, for both data rich and data-poor conditions. Additionally, it is recommended that the effect of habitat loss on the stock should be considered, although no clear targets have been set here. Development of targets for these habitat-related factors (not related to exploitation) is an option.

8.3 Development of the required knowledge base and methodology

Management options discussed in this report primarily refer to whole-stock conservation limits which need to be translated into appropriate local-system targets. Local management will depend on the locally available knowledge. However, several aspects of the biology of eel and several methodologies are currently inadequately understood to enable development of local management schemes. Co-ordinated research and development will facilitate local management. This should comprise:

- analysis of density-dependent processes (growth and mortality) and their impact on spawner escapement;
- quantification of the (positive) impacts of management measures not directly related to exploitation, e.g. habitat restoration, fish passes, re-stocking, etc.;
- development of harvest rate models for eel fisheries in data-rich systems;
- development of proximate criteria for management of fisheries in data-poor systems;
- development of procedures to post-evaluate potential effects of eel fisheries management measures, in both data-rich and data-poor systems.

8.4 The way ahead

Coherent research plans focusing on stock-wide management of the European eel have been prepared before (EIFAC, 1993; Moriarty and Dekker, 1997; ICES, 2000), but their impact on actual monitoring and research programmes has been marginal. The Terms of Reference for the current (2001) meeting of the Working Group on Eels allowed for consideration of a broad range of issues, to facilitate improving the scientific basis for advice. However, due to the lack of a co-ordinated management framework and the low priority of

national and local research programmes on eel, lack of progress has been reported on several issues, while others had to report only marginal progress. In addition, concerns have been raised about the ability to maintain existing monitoring efforts and time-series of data. Consequently, the Working Group has had to express its views on further development of the advice, without having the opportunity to fully address the Terms of Reference. Thus, some pragmatic reduction in the scope for development of further advice would appear to be appropriate in setting the Terms of Reference for coming meetings. However, cutting the coat to the cloth should not be read as an implicit statement that all management requests for advice could be fulfilled within such a pragmatically reduced setting.

A judicious choice for a feasible workload could include:

- Development of harvest rate models, including the derivation of exploitation levels corresponding to pre-set escapement targets and including the derivation of less data-demanding proximate criteria;
- The analysis of density dependent processes (growth, mortality and migration) and their effect on the production of escaping spawners;
- The analysis of habitat loss and the derivation of management goals for habitat restoration.

9 CONCLUSIONS AND RECOMMENDATIONS

9.1 Conclusions

Review of the available information on the status of the stock and fisheries of the European eel supports the view that stock is in decline in most of the distribution area and that fisheries is outside safe biological limits. Evidence has been given that anthropogenic factors (exploitation, habitat loss, increased predation, contamination and transfer of parasites and diseases) as well as natural processes (climate change) have contributed to the decline. Latest recruitment data (spring 2001) are indicative of further deterioration of the status of the stock.

The European eel stock extends through Europe and northern Africa and fisheries are scattered over many large and small water bodies. Management at the local level has failed to address the global decline of the stock, while effective management measures to restrict exploitation and to enhance the state of the stock are available.

Current scientific knowledge is inadequate to derive management targets specific for eel. However, anthropogenic impacts have been shown to exceed reasonable provisional targets in many places and management actions in compliance with provisional targets have been specified. Considering the many uncertainties and the uniqueness the eel stock (supposedly single panmictic, spawning only once in their lifetime), a precautionary reference point must ultimately be more strict than the universal reasonable first estimate (30 percent; ICES, 1997).

Noting the continuation of the decline in most recent recruitment indices, implementation of an international stock recovery plan is of utmost urgency.

9.2 Recommendations

The EIFAC/ICES Working Group on Eels at its 2001 session in Copenhagen (Denmark) recommends that:

- An international commission for the management of the European eel stock be formed, organizing monitoring and research on eel stocks and fisheries, serving as a clearing

house for regular exchange of information regarding landings and resource status, and facilitating and co-ordinating management action;

- A recovery plan for the eel stock be compiled and implemented as a matter of urgency and that fishing mortality be reduced to the lowest possible level until such a plan is agreed upon and implemented;
- A provisional limit reference point be set at an escapement from currently available habitat of female silver eel of at least 30 percent relative to the unexploited state, to be achieved by exploitation regulations and/or habitat restoration measures;
- Monitoring of recruitment, stocks, fisheries and escapement at least be sustained at recent levels, until a stock recovery plan is agreed upon and implemented, including a comprehensive monitoring and research plan.

10 REFERENCES

Aprahamian, M.W. 1988. Age structure of eel, *Anguilla anguilla* (L.), population in the River Severn, England, and the River Dee, Wales. *Aquaculture and Fisheries Management*, 19:365-376.

Avise, J.C., Helfman, G.S., Saunders, N.C. & Hales, L.S. 1986. Mitochondrial DNA differentiation in North Atlantic eels: population genetics consequences of an unusual life history pattern. *Proceedings of the National Academy of Sciences, USA*, 83:4350-4354.

Avise, J.C., Nelson,W.S., Arnold, J., Koehn, R.K., Williams, G.C. & Thorstelnsson, V. 1990. The evolutionary genetic status of Icelandic eels. *Evolution*, 44:1254-1262.

Baisez, A., Rigaud, C. & Feunteun, E. 2000. Hétérogénéité de répartition de l'anguille européenne (*A. anguilla*) dans un marais endigué de la côte atlantique (marais breton). *Cybium*, 34(3):23-32.

Barak, N.A.E. & Mason, C.F. 1992. Population density, growth and diet of eels *Anguilla anguilla* L., in two rivers in eastern England. *Aquaculture and Fisheries Management*, 23:59-97.

Barbin, G.P. & McCleave, J.D. 1997. Fecundity of the American eel *Anguilla rostrata* at 45° N in Maine, USA. *Jounal of Fish Biology*, 5:840-847.

Barral M. 2001. Etude préliminaire à la mise en place d'un "Tableau de bord anguille" sur le bassin Rhône-Méditerranée-Corse. Arles, France, Association Migrateurs Rhône-Méditerranée, 112 p.

Berg, S. & Jørgensen, J. 1994. Stocking experiments with 0+ eels (*Anguilla anguilla*) in Danish streams: post-stocking movements, densities and mortality. *In* I.G. Cowx, ed., *Rehabilitation of Freshwater Fisheries*, Oxford, Fishing News Books, pp. 314-325.

Boisneau, P. & Mennesson-Boisneau, C. 2001. Faisabilité de la mise au point d'un indice d'abondance de l'anguille d'avalaison en Loire. Paper presented at Journées anguilles du Groupe de Recherche Interdisciplinaire sur les Poissons Amphihalins (GRISAM), 26-29 Mars 2001, Tour du Valat, Arles, France.

Briand, C. & Fatin, D. 1999. Estuarine and fluvial recruitment of European glass eel in an Atlantic exploited estuary. Paper presented at 11[th] Session of the EIFAC/ICES Working Group on Eel, Silkeborg, Denmark, 20-24 September 1999.

Briand C., Feunteun, E., Fatin, D. & Fontenelle, G. 2000a. Assessment of mark recapture methods for glass eels (*Anguilla anguilla*) in Vilaine estuary, France. Paper presented at the International congress on eel research North American Fishery Society. Saint Louis, Missouri, United States

Briand C., Feunteun, E., Fatin, D. & Fontenelle, G. in press a. Development of a validated stock recruit model (*Anguilla anguilla*) in Vilaine river, France. *American Fisheries Society Symposium*

Briand, C., Fatin, D., Mounaix, B. & Feunteun, E. in press b. Mortalité des civelles en fonction de leur date d'arrivée en estuaire de la Vilaine. *Bulletin Français de la Pêche et de la Pisciculture*

Castonguay, M., Hodson, P., Moriarty, C., Drinkwater, K. & Jessop, B. 1994. Is there a role of ocean environment in American and European eel decline? *Fisheries Oceanography*, 3(3):197-203.

Ciccotti, E. 1997. Italy. *In* C. Moriarty & W. Dekker, eds, 1997. Management of the European Eel. *Fisheries Bulletin (Dublin)*, (15):91-100.

Dahl, J. 1967. Some recent observations on the age and growth of eels. In *Proceedings of the Third British Coarse Fish Conference*. Liverpool, University of Liverpool, pp. 48-52.

De Casamajor M.-N., Bru, N., Prouzet, P. & Sanchez, F. 2001. Suivi du recrutement de civelles (*Anguilla anguilla* L.) dans l'estuaire de l'Adour. *In* Océanographie du golfe de Gascogne. VIIème colloque international Biarritz, 4-6 avril 2000, edited by J. d'Elbée & P. Prouzet. *Actes de Colloques IFREMER*, 31:346-354.

Dekker, W. 1999. Is the European eel overfished? Paper presented at the 11th Session of the EIFAC/ICES Working Group on Eels, Silkeborg, Denmark, 20-24 September 1999.

Dekker, W. 2000a. The fractal geometry of the European eel stock. *ICES Journal of Marine Science*, 57:109-121.

Dekker, W. 2000b. A Procrustean assessment of the European eel stock. *ICES Journal of Marine Science*, 57:938-947.

Dekker, W. 2000c. Impact of yellow eel exploitation on spawner production in Lake Ijsselmeer, the Netherlands. *Dana*, 12:17-32.

Egusa, S. 1979. Notes on the culture of the European eel (*Anguilla anguilla* L.) in Japanese eel-farming ponds. *Rapports et Proces-Verbaux des Reunions, Conseil International pour l'Exploration de la Mer*, 174:51-58.

EIFAC 1993. Report of the Eighth Session of the Working Party on Eel, Olsztyn, Poland, 24-29 May 1993. *EIFAC Occasional Paper*, (27):21 p.

EIFAC/ICES 2001. Report of the Eleventh Session of the Joint EIFAC/ICES Working Group on Eels, Silkeborg, Denmark, 20-24 September 1999. *EIFAC Occasional Paper*, (34):31 p.

Environment Agency 2001. National Eel Management Strategy. Bristol, Environment Agency, 16 p.

Eybert, M.C, Bernard, J.Y., Constant, P., Feunteun, E., Hédin, J. & Questiau, S. 1999. Marsh meadow restoration in the Brière marsh (France): evolution of flora, birds and fishes. *Gibier Faune Sauvage*, 15:999-1016.

FAO 1996. Precautionary approach to capture fisheries and species introductions. *FAO, Technical Guidelines for Responsible Fisheries*, (2):52 p.

Feunteun, E., Rigaud, C., Elie, P. & Lefeuvre, J.C. 1992. Le marais doux endigué de Bourgneuf Machecoul (Pays de Loire). Premiers éléments de connaissance de la faune piscicole. Relations ichtyofaune-habitat et problèmes majeurs de gestion. *Revue des Sciences de l'Eau*, 5:509-528.

Feunteun, E, Acou, A., Guillouët, J., Laffaille, P. & Legault, A. 1998. Spatial distribution of an eel population (*Anguilla anguilla* L.) in a small coastal catchment of northern Brittany (France). Consequences of hydraulic works. *Bulletin Français de la Pêche et de la Pisciculture*, 349:129-139.

Feunteun, E., Rigaud, C., Elie, P. & Lefeuvre, J.C. 1999 Les peuplements piscicoles des marais littoraux de l'ouest de la France : un patrimoine à gérer ? *Bulletin Français de la Pêche et de la Pisciculture*, 352:63-79.

Feunteun, E., Acou, A., Laffaille, P. & Legault, A. 2000a. The European eel (*Anguilla anguilla*, L.): prediction of spawner escapement from continental population parameters. *Canadian Journal of Fisheries and Aquatic Science*, 57:1627-1635.

Feunteun, E., Boullier, J., Briaudet, J. & Laffaill, P. 2000b. La population d'anguille du Rhône aval: étude préliminaire en vue de l'élaboration d'un protocole de suivi et de restauration. Rennes, France, Direction Régional de l'Environnment Rhône Alpes, Electricité de France - Centre Nucléaire Production Electrique de Saint Alban, et Université de Rennes 1, 114 p.

Fournier, D. & Caron, F. 2001. Travaux de recherche sur l'anguille d'Amérique (Anguilla rostrata) de la Petite Rivière de la Trinité en 1999 et 2000. Quebec, Canada, Societé de la faune et des parcs du Québec, Direction de recherche sur la faune.

Francis, R.I.C.C. & Jellyman, D.J. 1999. Are mean size data adequate to monitor freshwater eel fisheries? *Marine and Freshwater Research*, 50:355-366.

Hilborn, R. & Walters, C.J. 1992. Quantitative Fisheries stock assessment: choice, dynamics and uncertainty. New York, Chapman and Hall, 570p.

ICES 1988. European eel assessment working group report, September 1987. Doc. ICES C.M. 1988/Assess:7, Copenhagen, Denmark, ICES.

ICES 1997. Report of the Study Group on the precautionary approach to fisheries management. Doc. ICES 1997/Assess:7, Copenhagen, Denmark, ICES

ICES 2000. Report of the ICES Advisory Committee on Fisheries Management, 1999, Part 1. *ICES Cooperative Research Report*, (236.1): 416 p.

ICES 2001. Report of the EIFAC/ICES Working Group on eels, St Andrews, N.B., Canada, 28 August - 1 September 2000. Doc. CM2001/ACFM:03. Copenhagen, ICES, 87 p.

Jones, T.A. & Hughes, J.M.R. 1993. Wetland inventories and wetland inventory studies: a European perspective. *In* M. Moser, R.C. Prentice & J. van Vessem, eds, *Waterfowl and wetland conservation in the 1990s: a global perspective. Proceedings of the IWRB Symposium, St. Petersburg, Florida, November 1992.* Wetlands International Publication No. 26.

Kamstra, A. 1999. Haltung, Fütterung und Aufzucht. *In* F.W. Tesch, *Der Aal*. London, Chapman and Hall, pp. 289-299.

Klein Breteler, J.G.P. 1994. Stockings of pre-winter and post-winter glass eels and of elvers from aquaculture: growth and survival in mesocosms. Paper presented at the 1994 Session of the EIFAC Working Party on Eels, Oviedio, Spain, 26 September - 1 October 1994.

Knights, B. 1997. Risk assessment and management of contamination of eels (*Anguilla* spp.) by persistent xenobiotic organochlorine compounds. *Chemistry and Ecology*, 13:171-212.

Knights, B. & White, E.M. 1997. An appraisal of stocking strategies for the European eel, *Anguilla anguilla* L. *In* I.G. Cowx, ed., *Stocking and Introduction of Fish*. Oxford, Fishing News Books, pp 121-140.

Knights B. & White, E.M. 1998. Enhancing immigration and recruitment of eels: the use of passes and associated trapping systems. *Fisheries Management and Ecology*, 5:459-471.

Knights B., Bark, A., Ball, M., Williams, F., Winter, E. & Dunn, S. 2001. Eel and elver stocks in England and Wales – Status and management options. R&D Technical Report No. W248. Bristol, United Kingdom, Environment Agency, 294 p.

Knösche, R., Zahn, S. & Borkmann, I. 2000. Fischereiliche Betroffenheitsanalyse zur Wasserkraftnutzung in Sachsen-Anhalt. Research report. Potsdam-Sacrow, Germany, Institute of Inland Fisheries, 76 p.

Lande, R., Sæther, B.-E. & Engen, S. 1997. Threshold harvesting for sustainability of fluctuating resources. *Ecology*, 78:1341-1350.

Legault, A. 1994. Preliminary study of the fluvial recruitment of eels. *Bulletin Français de la Pêche et de la Pisciculture*, 335:33-41.

Legault, A., Acou, A., Guillouët, J. & Feunteun, E. in press. Suivi de la migration d'avalaison d'anguilles dans une conduite de débit réservé. *Bulletin Français de la Pêche et de la Pisciculture*

Leopold, M. 1976. Efficiency and perspectives of eel management - summary and some economic approaches. Paper submitted to the Joint ICES/EIFAC Symposium on Eel Research and Management, Helsinki, Finland, 9-11 June 1976, Doc. 17, 17 p.

Lorenzen, K. 1996. The relation between body weight and natural mortality in juvenile and adult fish: a comparison of natural ecosystems and aquaculture. *Journal of Fish Biology*, 49:627-647.

Mace, P.M. 1994. Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. *Canadian Journal of Fisheries and Aquatic Science*, 51:110-122.

Maes, G. & Volckaert, F. 1999. Genetic structure of (sub)adult European eel based on allozyme markers. Paper presented at the 11th Session of the EIFAC/ICES Working Group on Eels, Silkeborg, Denmark, 20-24 September 1999.

Mann, R.H.K. 1995. Eel stock assessment in the UK. Report to MAFF, Project No. SF0305, IFE Project No.T11063al. London, Ministry of Agriculture, Fisheries and Food, London, 86 p.

McCarthy, T.K., Allison, J., Callaghan, R., Purcell, P. & Copely, L. 1994a. Yellow eel stocks in the River Shannon and the development of the fyke net fishery. Report to the Electricity Supply Board, Dublin, Ireland.

McCarthy, T.K., Cullen, K., Faherty, K. & O'Farrell, M.M. 1994b. River Shannon silver eels: population biology, factors influencing downstream migration and commercial fishing. Report to the Electricity Supply Board, Dublin, Ireland.

McGurk, M.D. 1996. Allometry of marine mortality of Pacific salmon. *Fisheries Bulletin*, 94:77-88.

Moriarty, C. 1987. The eel stocks of the Shannon system and prospects for the development of the fishery. Fishery Leaflet No. 139. Dublin, Department of the Marine, 8 p.

Moriarty, C. 1990. Eel management practice in three lake systems in Ireland, *In* W.L.T. van Densen, B. Steinmetz & R.H.Hughes, eds, *Management of Freshwater Fisheries. Proceedings of a Symposium organised by EIFAC, Göteborg, Sweden, 31 May - 3 June 1988*. Wageningen, Netherlands, Pudoc, pp. 262-269.

Moriarty, C. & Dekker, W. (eds) 1997. Management of the European eel – Enhancement of the European eel fishery and conservation of the species. *Fisheries Bulletin (Dublin)*, (15):110 p.

Moriarty, C. & McCarthy, D. 1982. Eel. *In* EIFAC, Report of the symposium on stock enhancement in the management of freshwater fisheries. *EIFAC Technical Paper*, (42):3-6.

Mower, D. 1998. Leighton Moss elver study, 1997-8. Unpublished report. Leighton Moss, United Kingdom, Royal Society for the Protection of Birds.

Mower, D. 1999. Elver study at Leighton Moss 1999 in relation to commercial elver fishing. Unpublished report. Leighton Moss, United Kingdom, Royal Society for the Protection of Birds.

Naismith, I.A. & Knights, B. 1988. Migrations of elver and juvenile European eel, *Anguilla anguilla* L., in the River Thames catchment. *Journal of Fish Biology*, 42:217-226.

Naismith I.A. & Knights, B. 1993. The distribution, density and growth of European eel, *Anguilla anguilla* L., in the River Thames. *Journal of Fish Biology*, Suppl.A:161-175.

Nicola, G.G., Elvira, B. & Almodóvar, A. 1996. Dams and fish pasaje facilities in the large rivers of Spain: effects on migratory species. *Acta Hydrobiologica*, Suppl.152: 375-379.

Pedersen, M.I. 1997. Denmark. *In* C. Moriarty & W. Dekker, eds, Management of the European Eel. *Fisheries Bulletin (Dublin)* (15):53-56.

Pedersen, M.I. 2000. Long-term survival and growth of stocked eel, *Anguilla anguilla* (L.) in a small eutrophic lake. *Dana*, 12:71-76.

Pedersen, M.I. & Dieperink, C. 2000. Fishing mortality on silver eels, *Anguilla anguilla* (L.), in Denmark. *Dana*, 12:77-82.

Pursiainen M. & Toivonen, J. 1984. The enhancement of eel stocks in Finland; a review of introduction and stockings. *EIFAC Technical Paper*, (42, Suppl.1):59-67.

Rasmussen, G. & Therkildsen, B. 1979. Food, growth and production of *Anguilla anguilla* L. in a small Danish stream. *Rapports et Proces-Verbaux des Reunions, Conseil International pour l' Exploration de la Mer*, 174:115-123.

Robinet, T. & Feunteun, E. 2002. Sublethal effects of exposure to chemical compounds: a cause for the decline in Atlantic eels? *Ecotoxicology*, 11:265-277.

Rosenberg, A., Mace, P., Thompson, G., Darcy, G., Clark, W., Collie, J., Gabriel, W., MacCall, A., Methot, R., Powers, J., Restrepo, V., Wainwright, T., Botsford, L., Hoenig, J. & Stokes, K. 1994. Scientific review of definitions of overfishing in U.S. Fishery Management Plan. Silver Spring MD, USA, U.S. National Oceanic and Atmospheric Administration. NOAA Tech. Memor. NMFS-F/SPO-17, 205p.

Sparre, P. 1979. Some necessary adjustments for using the common methods in eel assessment. *Rapports et Proces-Verbaux des Reunions, Conseil International pour l'Exploration de la Mer*, 174:41-44.

Svärdson, G. 1976. The decline of the Baltic eel population. *Report. Institute of Freshwater Research, Drottningholm*, 55:136-143.

Svedäng, H. 1999. Vital population statistics of the exploited eel stock on the Swedish west coast. *Fisheries Research*, 40:251-265.

Turnpenny, A.W.H. 1989. Field studies on fisheries in acid waters in the United Kingdom. *Society for Experimental Biology Seminar Series*, (34):45-65.

Valente, A. 1993. Impactes de emprendimentos hidroeléctricos sobre as comunidades piscícolas. *Recursos Hídricos*, 13(3): 9-14.

Vøllestad L.A. 1986. Growth and production of female yellow eels (*Anguilla anguilla* L.) from brackish water in Norway. *Vie et Milieu, Paris*, 36(4):267-271.

Vøllestad L.A. & Jonsson, B. 1988. A 13-year study of the population dynamics and growth of the European eel *Anguilla anguilla* in a Norwegian river: evidence for density-dependent mortality and development of a model for predicting yield. *Journal of Animal Ecology*, 57:983-997.

Walters, C. & Maguire, J.-J. 1996. Lessons for stock assessment from the northern cod collapse. *Reviews in Fish Biology and Fisheries*, 6:125-137.

Westin, L. 1998. The spawning migration of European silver eel (Anguilla anguilla L.) with particular reference to stocked eel in the Blatic. *Fisheries Research*, 38(3):257-270.

White, E.M. & Knights, B. 1994. Elver and eel stock assessment in the Severn and Avon. NRA R&D Project Record 256/13/ST. Bristol, United Kingdom, National Rivers Authority, 141 p.

White, E.M. & Knights, B. 1997a. Dynamics of upstream migration of the European eel, *Anguilla anguilla* (L.), in the Rivers Severn and Avon, England, with special reference to the effects of man-made barriers. *Fisheries Mangement and Ecology*, 4:311-324.

White, E.M. & Knights, B. 1997b. Environmental factors affecting the migration of the European eel in the Rivers Severn and Avon, England. *Journal of Fish Biology*, 50:1104-1116.

Wickström, H. & Hamrin, S. 1997. Sweden. *In* C. Moriarty & W. Dekker, eds, 1997. Management of the European Eel. *Fisheries Bulletin (Dublin)*, (15):46-52.

Wickström, H., Westin L. & Clevestam, P. 1996. The biological and economical yield from a long-term eel stocking experiments. *Ecology of Freshwater Fish*, 5:140-147.

Wirth, Th. & Bernatchez, L. 2001. Genetic evidence against panmixia in the European eel. *Nature*, 409:1037-1040.

Table I A

Recruitment data series. In this table, recruitment data series are listed, in the units in which they were reported. Part 1: Scandinavia and British Isles

	N	S	S	S	S	Dk	D	N.Irl	Irl	Irl	UK
	Imsa	Viskan	Upsala	Motala	Göta Älv	Vidaa	Ems	Bann	Erne	Shannon	Severn
1950				305	2947		875				
1951			210	2713	1744		719				
1952			324	1544	3662		1516				
1953			242	2698	5071		3275				
1954			509	1030	1031		5369				
1955			550	1871	2732		4795		0.2		
1956			215	429	1622		4194				
1957			162	826	1915		1829				
1958			337	172	1675		2263				
1959			613	1837	1745		4654		0.2		
1960			289	799	1605		6215	7409	1.2		
1961			303	706	269		2995	4939	0.6		
1962			289	870	873		4430	6740	2.5		
1963			445	581	1469		5746	9077	0.4		
1964			158	182	622		5054	3137	0.2		
1965			276	500	746		1363	3801	0.9		
1966			158	1423	1232		1840	6183	1.4		
1967			332	283	493		1071	1899	0.3		
1968			266	184	849		2760	2525	1.5		
1969			34	135	1595		1687	422	0.6		
1970			150	2	1046		683	3992	0.1		
1971		12	242	1	842	787	1684	4157	0.5		
1972		88	88	51	810	780	3894	2905			
1973		177	160	46	1179	641	289	2524			
1974		13	50	59	631	464	4129	5859	0.8		
1975		99	149	224	1230	888	1031	4637	0.4		
1976		500	44	24	798	828	4205	2920	0.4		
1977		850	176	353	256	91	2172	6443	0.1	1.0	
1978		533	34	266	873	335	2024	5034	0.3	1.4	
1979		505	34	112	190	220	2774	2089	0.5	6.7	40
1980		72	71	7	906	220	3195	2486	1.4	4.5	33
1981		513	7	31	40	226	962	3023	2.3	2.1	32
1982		380	1	22	882	490	674	3854	4.4	3.2	30
1983	7	308	56	12	113	662	92	242	0.7	6.3	6
1984	3	21	34	48	325	123	352	1534	1.1	5.1	29
1985		200	70	15	77	13	260	557	0.4	1.1	19
1986		151	28	26	143	123	89	1848	0.7	0.9	16
1987	2	146	74	201	168	341	8	1683	2.3	1.6	18
1988	7	92	69	170	475	141	67	2647	3.0	0.1	23
1989	4	32		35	598	9	13	1568	1.7	0.1	14
1990	13	42		21	149	5	99	2293	2.2	0.5	16
1991	3	1		2	264		52	677	0.5	0.1	8
1992	2	70	8	108	404		6	978	1.4	0.1	18
1993	3.4	43	6	89	64		20	1525	1.8	0.1	21
1994	0.2	76	72	650	377		52	1249	3.5	0.3	22
1995	0.8	6	8	32			40	1403	2.4	0.4	36
1996	0.4	1	18	14	277		20	2667	1.0	0.5	24
1997		8		8	180		5	2533	1.0	2.0	17
1998		5		6			4	1283	0.8	0.1	20
1999		2		85			3	1345	1.1	0.1	18
2000		14		270			4	563	0.9		8
2001		2					0	250	0.7		

Table I B

Recruitment data series; continued. Part 2: Mainland Europe

	NL Den Oever	B Yser	F Loire	F Vilaine	F Gironde (CPUE)	F Gironde (Yield)	F Adour	E Nalon	P/E Minho	It Tiber
1950	7		86							
1951	13		166							
1952	84		121							
1953	12		91							
1954	18		86							
1955	25		181					14		
1956	7		187					17		
1957	15		168					15		
1958	48		230					14		
1959	27		174					13		
1960	21		411					19		
1961	36		334					13		
1962	80		185					18		
1963	115		116					11		
1964	36	4	142					16		
1965	75	115	134					20		
1966	18	385	253					12		
1967	28	575	258					13		
1968	19	554	712					22		
1969	16	445	225					16		
1970	36	795	453					198		
1971	17	399	330					18		
1972	29	557	311	39				11		
1973	22	356	292	78				11		
1974	25	946	563	107				25	2	
1975	32	264	495	44				32	11	11.0
1976	26	618	770	106				55	20	6.7
1977	57	450	654	52				37	37	5.9
1978	37	388	523	105				650	24	3.6
1979	50	675	608	209	20	286		77	28	8.4
1980	26	358	502	95	26	405		42	21	8.2
1981	22	74	284	57	20	332		35	54	4.0
1982	14	138	266	98	15	123		27	16	4.0
1983	9	10	276	69	14	80		22	30	4.0
1984	12	6	168	36	19	82		23	31	1.8
1985	14	13	159	32	10	65		12	21	2.5
1986	14	26	137	48	11	45	8	14	13	0.2
1987	6	33	93	32	14	82	10	24	8	7.4
1988	4	48	138	39	11	33	12	15	8	10.5
1989	3	30	61	30	7	80	9	14	9	5.5
1990	3	218	76	31	6	48	3	9	6	4.4
1991	1	13	30	15	8	64	2	7	9	0.8
1992	3	19	32	30	4	42	8	11	10	0.6
1993	3	12	80	32	8		4	10	8	0.5
1994	4	18	95	24	9		3	10	5	0.5
1995	7	2	68	30	8		8			0.3
1996	7	5	32	22	5		4			0.1
1997	12	10	90	23	7		5			0.1
1998	2	8		18	4		2			0.1
1999	3	76		15			4			0.1
2000	2			14			9			
2001	0.5			8						

Table II

Statistics of eel landings, reported in the FAO database of fishing yields.
These data include landings of 'river eels' in Atlantic waters, the Mediterranean and Inland waters.
Data for Denmark, Netherlands and Italy have been corrected for incorrectly included aquaculture yield

	Norway	Sweden	Denmark	Germany	Ireland	UK	Nether-lands	France	Spain	Portu-gal	Italy	Rest Europe	N. Africa
1950	300	2200	4500	400			4200	500	100		1000		
1951	300	1900	4400	400			3700	500	100		1000		
1952	200	1600	3900	400			4000	700	100		1000		
1953	400	2400	4300	500		400	3100	600	100		1000	900	
1954	300	2100	3800	300		500	2100	500	900		1000	800	
1955	500	2600	4800	500		700	1700	500	600		1000	1000	
1956	300	1500	3700	400		600	1800	500	800		2000	900	
1957	400	2200	3600	400		600	2500	500	500		2000	800	
1958	400	1800	3300	400	100	600	2700	500	500		2100	1200	
1959	400	2800	4000	500	100	500	3400	900	500		3000	700	
1960	400	1500	4723	400	0	800	3000	1300	500		2700	1000	
1961	500	2100	3875	500	100	800	2660	1300	400		2600	900	300
1962	400	1900	3907	400	100	700	1543	1300	800		3100	1000	300
1963	500	1900	3928	2100	100	700	1818	1400	1100		3500	1000	300
1964	400	2368	3282	1900	100	600	2368	1400	1700		3500	1100	400
1965	500	1868	3197	1500	200	800	2509	1700	1300		3200	900	500
1966	500	2070	3690	1700	100	1000	2739	1300	1300		3100	1000	400
1967	500	1667	3436	1900	100	600	2884	2000	1400		3100	1100	400
1968	600	1872	4218	1800	100	600	2622	2700	1300		3200	1100	400
1969	500	1773	3624	1600	100	600	2741	1900	1400		3400	1100	400
1970	400	1270	3309	1600	200	800	1512	4200	1100		3300	1400	100
1971	400	1469	3195	1300	100	800	1153	4900	1100		3400	1500	100
1972	400	1274	3229	1300	100	700	1057	2600	1000		2900	1138	100
1973	400	1277	3455	1300	100	800	1023	3900	700		2900	1150	800
1974	383	1106	2814	1285	67	817	994	2493	1300	42	2697	1528	352
1975	411	1492	3225	1398	79	833	1173	1590	570	44	2973	1400	85
1976	386	1023	2876	1322	150	694	1306	2959	675	38	2677	1254	47
1977	352	1084	2323	1317	108	742	929	1538	666	52	2462	1384	159
1978	347	1162	2335	1162	76	877	862	2455	655	44	2237	1357	112
1979	374	1043	1826	1164	110	879	687	3144	394	25	2422	1518	134
1980	387	1205	2141	1051	75	1053	828	4503	300	32	2264	1242	448
1981	369	976	2087	1033	94	858	876	1425	250	33	2340	1192	497
1982	385	1250	2378	1027	144	1032	1097	1469	200	14	2087	1419	455
1983	324	1304	2003	1029	117	1113	1230	1856	150	11	2076	1782	575
1984	309	1176	1745	911	88	957	681	2336	150	80	2361	2445	477
1985	352	1261	1519	866	87	781	666	2288	200	76	1907	2123	258
1986	271	981	1552	887	87	997	729	2924	200	633	1928	1867	356
1987	282	896	1189	731	221	939	512	2378	259	566	2076	2479	306
1988	513	1198	1759	746	215	715	590	2879	205	501	2165	2790	256
1989	312	1141	1582	678	400	1075	645	2482	83	6	1301	2365	368
1990	336	1120	1568	976	256	1039	657	2484	75	295	1199	2209	560
1991	323	1244	1366	1010	245	822	707	2260	65	314	1106	2337	358
1992	373	1375	1342	1026	234	782	621	1964	60	674	1662	2749	358
1993	340	1336	1023	1027	260	752	320	1674	55	505	1307	2509	613
1994	472	1480	1140	585	300	873	369	1417	50	979	986	2797	732
1995	454	1257	840	585	400	808	279	500	106	10	886	2572	1176
1996	352	1226	717.5	696	550	895	336	563	97	21	883	2676	984
1997	497	1288	757.6	746	550	807	315	1942	113	16	1010	2034	1327
1998	353	877	557	717	670	741	346	491	160	13	682	2159	1069
1999	475	987	686	747	675	697	372	189	166	3		1532	1257
2000							368						

Table III

Re-stocking of glass eel. Numbers of glass eels (in millions) re-stocked in (eastern) Germany (D east), the Netherlands (NL), Sweden (S), Poland (PO) and Northern Ireland (N.Irl.)

	D (east)	N	S	PO	North. Irl.	SUM
1945					17.0	17.0
1946		7.3			21.0	28.3
1947		7.6				7.6
1948		1.9				1.9
1949		10.5				10.5
1950	0.0	5.1				5.1
1951	0.0	10.2	0.0			10.2
1952	0.0	16.9	0.1	17.6		34.5
1953	2.2	21.9	0.0	25.5		49.6
1954	0.0	10.5		26.6		37.1
1955	10.2	16.5		30.8	0.5	58.0
1956	4.8	23.1		21.0		48.9
1957	1.1	19.0		24.7		44.8
1958	5.7	16.9		35.0		57.6
1959	10.7	20.1		52.5	0.7	83.9
1960	13.7	21.1		64.4	25.9	125.1
1961	7.6	21.0		65.1	16.7	110.4
1962	14.1	19.8		61.6	27.6	123.1
1963	20.4	23.2		41.7	28.5	113.8
1964	11.7	20.0	0.0	39.2	10.0	80.9
1965	27.8	22.5		39.8	14.2	104.4
1966	21.9	8.9		69.0	22.7	122.6
1967	22.8	6.9		74.2	6.7	110.7
1968	25.2	17.0			12.1	54.3
1969	19.2	2.7			3.1	25.0
1970	27.5	19.0			12.2	58.6
1971	24.3	17.0			14.1	55.4
1972	31.5	16.1			8.7	56.3
1973	19.1	13.6			7.6	40.2

	D (east)	N	S	PO	North.. Irl.	SUM
1974	23.7	24.4			20.0	68.1
1975	18.6	14.4			15.1	48.1
1976	31.5	18.0			9.9	59.5
1977	38.4	25.8			19.7	83.9
1978	39.0	27.7			16.1	82.8
1979	39.0	30.6	0.1		7.7	77.5
1980	39.7	24.8	0.1		11.5	76.1
1981	26.1	22.3			16.1	64.5
1982	30.6	17.2			24.7	72.5
1983	25.2	14.1			2.9	42.2
1984	31.5	16.6			12.0	60.1
1985	6.0	11.8	0.8		13.8	32.3
1986	23.8	10.5	0.1		25.4	59.8
1987	26.3	7.9	0.0		25.8	59.9
1988	26.6	8.4	0.2		23.4	58.6
1989	14.3	6.8	0.0		9.9	31.0
1990	10.5	6.1	0.7		13.3	30.6
1991	1.9	1.9	0.3		3.5	7.6
1992	6.2	3.5	0.3		9.4	19.4
1993	7.6	3.8	0.6		9.9	21.9
1994	7.4	6.2	1.7		16.4	31.8
1995	6.2	4.8	1.5		13.5	26.0
1996	0.5	1.8	2.3		11.1	15.7
1997	0.4	2.3	2.4		10.9	16.1
1998	0.0	2.5	2.1		6.2	10.9
1999	0.0	2.9	2.2		12.0	17.1
2000		2.8	1.2		5.4	9.4
2001		0.9			2.8	3.7

Table IV

Re-stocking of bootlace eel. Numbers of bootlace eels (in millions) re-stocked in (eastern) Germany (D east), the Netherlands (NL), Sweden (S) and Denmark (DK)

	D (east)	NL	S	DK	SUM
1945					0.0
1946					0.0
1947		1.6			1.6
1948		2.0			2.0
1949		1.4	0.0		1.4
1950	0.9	1.6	0.0		2.5
1951	0.9	1.3	0.0		2.2
1952	0.6	1.2	0.0		1.8
1953	1.5	0.8	0.0		2.3
1954	1.1	0.7	0.0		1.8
1955	1.2	0.9	0.0		2.2
1956	1.3	0.7	0.0		2.0
1957	1.3	0.8	0.0		2.1
1958	1.9	0.8	0.0		2.8
1959	1.9	0.7	0.0		2.6
1960	0.8	0.4	0.0		1.2
1961	1.8	0.6	0.0		2.4
1962	0.8	0.4	0.0		1.2
1963	0.7	0.1	0.0		0.9
1964	0.8	0.3	0.1		1.3
1965	1.0	0.5	0.1		1.6
1966	1.3	1.1	0.1		2.5
1967	0.9	1.2	0.1		2.2
1968	1.4	1.0	0.1		2.5
1969	1.4	0.0	0.0		1.4
1970	0.7	0.2	0.0		1.0
1971	0.6	0.3	0.0		1.0
1972	1.9	0.4	0.1		2.4
1973	2.7	0.5	0.1		3.3
1974	2.4	0.5	0.1		3.0
1975	2.9	0.5	0.1		3.6
1976	2.4	0.5	0.1		2.9
1977	2.7	0.6	0.0		3.3
1978	3.3	0.8	0.1		4.2
1979	1.5	0.8	0.1		2.4
1980	1.0	1.0	0.1		2.1
1981	2.7	0.7	0.1		3.6
1982	2.3	0.7	0.4		3.4
1983	2.3	0.7	1.0		4.0
1984	1.7	0.7	0.8		3.2
1985	1.1	0.8	0.9		2.8
1986	0.0	0.7	0.5		1.2
1987	0.0	0.4	1.0	1.6	3.0
1988	0.0	0.3	1.3	0.8	2.4
1989	0.0	0.1	1.0	0.4	1.5
1990	0.1	0.1	1.6	3.5	5.3
1991	0.2	0.1	1.8	3.1	5.1
1992	0.2	0.0	2.2	3.9	6.3
1993	0.3	0.0	2.0	4.0	6.3
1994	0.4	0.1	2.0	7.4	9.9
1995	0.4	0.1	1.8	8.4	10.7
1996	0.9	0.0	2.5	4.6	8.1
1997	2.3	0.1	2.5	2.5	7.4
1998	1.8	0.1	2.4	3.0	7.3
1999	1.1	0.1	2.4	4.1	7.7
2000		0.0	1.5	3.8	5.3
2001				1.7	1.7

Table V

Inter- and intra-catchment re-stocking. Percentage intra-catchment re-stockings in the Netherlands (NL) and Northern Ireland (N.Irl.)

	NL	North. Irl.
1958	0.0	
1959	0.0	
1960	15.0	100
1961	14.3	100
1962	15.6	100
1963	12.7	100
1964	15.0	100
1965	11.9	100
1966	21.0	100
1967	15.2	100
1968	18.1	100
1969	52.3	100
1970	17.4	100
1971	na	100
1972	47.8	100

	NL	North. Irl.
1973	43.6	100
1974	44.2	100
1975	56.6	100
1976	23.7	100
1977	37.2	100
1978	33.0	100
1979	30.8	100
1980	38.4	100
1981	37.0	100
1982	10.9	100
1983	22.2	100
1984	7.6	53.5
1985	2.1	13.3
1986	7.4	23.7
1987	4.4	26.9

	NL	North. Irl.
1988	13.5	55.7
1989	7.9	100
1990	29.3	100
1991	50.9	100
1992	26.3	55.4
1993	15.9	100
1994	40.9	61.8
1995	10.7	67.2
1996	62.8	98.8
1997	66.7	97.3
1998	68.6	98.7
1999	46.6	53.5
2000	42.7	78.9
2001	27.3	100

Table VI

Production of European eel in aquaculture in Europe and Japan. Compilation of production estimates (tonnes) derived from reports of previous meetings, FAO, FEAP and others

	1984	1985	1986	1987	1988	1989	1990	1991	1992
Norway									
Sweden	15	47	59	193	233	190	160	195	179
Denmark	16	30	120	160	300	620	900	900	706
Germany									
Ireland									
UK				20	30	0	0		
Netherlands		20	100	200	200	350	550	520	500
Belgium/Lux.					30	30	125	125	30
Spain	15	20	25	37	32	57	98	105	130
Portugal	60	60	590	566	501	6	270	622	267
Marocco							35	41	60
Algeria					72	53	22	1	46
Tunisia							150	151	144
Italy	2600	2800	4200	4600	4250	4500	3700	4185	3265
Greece			6	4	10	54	94	132	81
Turkey									
Macedonia									
Yugoslavia	44	52	48	49	19	10	5	1	8
Croatia								7	
Hungary					90	39	73	33	98
Czech Republic									
sum EU	1950	2229	3448	4729	5517	5159	6667	6098	6349
Japan		3000							

	1993	1994	1995	1996	1997	1998	1999	2000	2001
Norway		120	200	200	200	200			
Sweden	192	182	158	184	215	250	250	250	260
Denmark	900	1000	1200	1200	1700	2468	2700	2675	2500
Germany		100	100	100	150	150	150	150	300
Ireland									100
UK		25		25					
Netherlands	1250	1487	1535	2800	2443	3250	3800	4000	3800
Belgium/Lux.	125	125	150	140	150	150	40	20	50
Spain	175	134	214	249	266	270	300	425	200
Portugal	505	979	200	110	200	200	200	200	
Marocco	68	85	55	55	56				
Algeria	0	22	20	17	17				
Tunisia	250	260	108	158	147	108			
Italy	3000	2800	3000	3000	3100	3100	3100	2800	2700
Greece	337	341	659	550	312	500	500	300	600
Turkey									
Macedonia	1	0	70	83	60				
Yugoslavia	8	2	9	5	5				
Croatia	5	5	7	6	7				
Hungary		50		50			19	19	
Czech Republic	2	4	4	3	3				
sum EU	6818	7721	7689	8935	9031	10646	11059	10839	10510
Japan									10000

Table VII

Natural and fishing mortality values for eel (M and F represent the instantaneous rate of natural and fishing mortality respectively; percentage mortality represents the mortality in percentage terms between time t and t+n, where n is variable and dependent on the study), adapted from Knights *et al.* (2001)

Habitat Location	Natural mortality %	Natural mortality M	Fishing mortality %	Fishing mortality F	Comments	Reference
River and Stillwaters – for combined yellow and silver eel						
Lough Neagh (Irl)	82% pre-fishery		75%		Stocked system heavily exploited	Knights and White, 1997 Moriarty and McCarthy, 1982
Shannon lakes (Irl)			20%		Low exploitation rates	McCarthy *et al.*, 1994a, Moriarty, 1987; 1990
L.Derg andRee (Irl.)	68-74%	0.3-0.38			Low exploitation rates	McCarthy *et al.*, 1994b
Swedish lake	<90%				Stocked in 1980, study not completed in 1994	Wickström, Westin and Clevestam, 1996
Finnish lakes	28-73%				Stocked, variable exploitation	Pursianen and Toivonen, 1984
Byelorussian lakes	Total mortality 95%				Stocked	Leopold, 1976
Ijsselmeer (NL)				1.0	Yellow and silver eels heavily exploited	Dekker, 2000b
Small lake (Dk)	24-45%	Z = 0.07-0.04			Stocking study after seven years-wild eel	Pedersen, 2000
Ponds (NL)	30-77%	0.17-0.65			Fingerling stocking experiments	Klein Breteler, 1994
Ponds (Dk)	89%				Stocking study after one year	Dahl, 1967
Coastal lagoons (It)		0.25-0.72			Commachio stocked lagoons	Ciccotti, 1997
Rivers in England. (UK)	75%				From age-cohort analyses	Barak and Mason, 1992
River Imsa (N)	50-84% (av. 73%)	0.088-0.225 (av. 0.167)			Recruitment and escapement measured over 13 years	Vøllestad and Jonsson, 1988
River Gudena (Dk)	77-82%				Post-stocking studies	Berg and Jørgensen, 1994
Stocked streams (Dk)	97%	0.23-1.79			Post-stocking studies	Pedersen, 1997
Unstocked streams (Dk)	60%				Post-stocking studies (still in progress)	Pedersen, 1997
Stream (Dk)		0.36-0.65			From age-cohort analyses	Rasmussen and Therkildsen, 1979
River Tiber (It)		0.8			From age-cohort analyses	Ciccotti, 1997
West Coast (S)		0.23	96.5%	0.16-0.42	Coastal fyke net fishery data	Svedäng, 1999
S.E. Coast(N)		0.04-0.46		0.02-0.08	Coastal fyke net fishery data	Vøllestad, 1986
German Bight				0.2	Age-structured cohort analysis	Sparre, 1979
Thames Estuary (UK)	>95%	Z = 0.5-0.7		Z = 0.5-0.7	Compartmental Z model, comparing exploited v. unexploited zones	Naismith and Knights, 1993

Table VII

(continued)

Habitat Location	Natural mortality %	M	Fishing mortality %	F	Comments	Reference
Glass eel fisheries						
Severn, England	>99%		<1.0%		Mark-recapture	Knights *et al.*, 2001
Creek, NW UK			>70-90%		Trapping study	Mower, 1998; 1999
Bay of Biscay (F)				3.15	Fishing mortality over the glass eel stage	Dekker, 2000b
Vilaine (F)			96-99%		Mark-recapture and trapping below barrage	Briand *et al.*, in press, b
Adour (F)			20-25%			De Casamajor *et al.*, 2001
Silver eel fisheries						
Coast (S)	80%	0.18	27%		Mark-recapture	Wickström *et al.*, 1996
Limfjord (Dk)			43-88%		Mark-recapture	Pedersen, 1997
Coast (Dk)			19-38%		Mark-recapture	Pedersen and Dieperink, 2000

Table VIII

Status of eel stocks with respect to impact of yellow eel fisheries. Table VII lists case studies, this table provides indications for country averages

	Principal Fisheries			**Level of Exploitation**			**Qualitative impact of fishery on spawner escapement**
	Glass eel	**Yellow eel**	**Silver eel**	**Low**	**Optimal**	**Over-exploited**	
Sweden West Coast		x			x		Predominantly yellow eel fishery. Low degree of spawner escapement?
Sweden East Coast		x	x		x		Mixed yellow and silver eel fishery. Low spawner escapement?
Denmark		x	x		x		Mixed yellow and silver eel fishery (50/50). Overall low spawner escapement from the Baltic Sea.
Germany		x	x		x		Mixed yellow and silver eel fishery. Moderate degree of spawner escapement.
Northern Ireland (Lough Neagh)		x	x		x		Predominantly yellow eel fishery, with significant silver eel catch. Significant proportion of spawner escapement.
UK	x			x			Significant local glass eel fishery. Scattered, low-intensity yellow and silver eel fisheries. Minimal impacts on stocks and spawner escapement (cf. Unexploited populations)
The Netherlands (IJsselmeer)		x				x	Low yield of silver eels. Fully fished yellow eel fishery. Very low spawner escapement
France	x	x		x			Large glass eel fishery, but some other mixed yellow and silver eel fisheries. Overall high degree of spawner escapement.
Spain	x			x			Mostly glass eel fishery. High degree of spawner escapement
Portugal	x					x?	Limited data available. Significant illegal fishery suspected.

Level of exploitation:	Low	$F \leq M$	fishing mortality equal or less than natural mortality
	Optimal	Fmax	maximum yield per recruit
	Over-exploited	$F \geq 1$	growth-overfishing evident

Table IX

Commonly used reference points and their associated data requirements (source: ICES, 1997)

Reference Point	Definition	Data requirements
	Fishing rates	
F0.1	Fishing mortality rate (F) at which slope of the yield per recruit (Y/R) curve is 10 percent of its value near the origin	Weight at age, natural mortality, exploitation pattern (F, partial recruitment vector)
Fmax	F giving the maximum yield on a Y/R curve	Weight at age, natural mortality, exploitation pattern (F, partial recruitment vector)
FMSY	F corresponding to Maximum Sustainable Yield from a production model or from an age-based analysis using a stock recruitment model	Weight at age, natural mortality, exploitation pattern (F, partial recruitment vector), stock recruitment relationship or general production models
F30%SPR	F corresponding to a Spawning Stock Biomass per Recruit (SSB/R) which is 30 percent of the SSB/R obtained when F = 0	Weight and maturity at age, natural mortality, exploitation pattern (F, partial recruitment vector)
F >= M	Empirical (for top predators)	M and sustainable F's for similar resources
Fcrash	F represented by the tangent through the origin of a stock recruitment relationship	Weight at age, natural mortality, exploitation pattern (F, partial recruitment vector), stock recruitment relationship
Fpa	F precautionary approach, used to constrain mortality to ensure high probability of exceeding Flim (mortality limit reference point)	Same as data required for other F reference point calculations
Biomass / Spawning Stock Levels		
BMSY	Biomass corresponding to Maximum Sustainable Yield from a production model or from an age-based analysis using a stock recruitment model	Weight at age, natural mortality, exploitation pattern (F, partial recruitment vector), stock recruitment relationship or general production models
MBAL	A value of SSB below which the probability of reduced recruitment increases	Data series of spawning stock size and recruitment
B50%R	The level of spawning stock at which average recruitment is one half of the maximum of the underlying stock recruitment relationship	Stock recruitment relationship
B20%B-virg	Level of spawning stock corresponding to a fraction (for ex. 20 percent) of the unexploited biomass	Weight at age, natural mortality, exploitation pattern (F, partial recruitment vector), stock recruitment relationship
Bpa	B precautionary approach, used as constraint on mortality to ensure a high probability of exceeding Blim (biomass limit reference point)	Same as data required for other B reference point calculations

Appendix

LIST OF PARTICIPANTS

Antunes, Carlos
Centro de Investigação Marinha
e Ambiental (CIMAR)
Rua dos Bragas, 177
P-4050-123 Porto
Portugal
Tel.: +351 22 340 18 08
Fax: +351 22 339 06 08
E-mail: cantunes@cimar.org

Aprahamian, Miran
Environment Agency
Northwest Region
Richard Fairclough House
Knutsford Road
Warrington WA4 1HG
United Kingdom
Tel.: +44 1925 653999
Fax: +44 1925 415961
E-mail: miran.aprahamian
@environment-agency.gov.uk

Cairns, David
Department of Fisheries and Oceans
Science Branch
Charlottetown, PEI C1A 7M8
Canada
Tel.: +1 902 5667825
Fax: +1 902 5667948
E-mail: cairns@dfo-mpo.gc.ca

Dekker, Willem (Chairman)
Netherlands Institute for
Fisheries Research
P.O.Box 68
Haringkade 1
NL-1970 AB Ijmuiden
The Netherlands
Tel.: +31 255 564712
Fax: +31 255 564644
E-mail: willem@rivo.dlo.nl

Feunteun, Eric
University of Rennes I
UMR CNRS 6553
Avenue du Général Leclerc
F-35042 Rennes
France
Tel.: +33 299 281439
Fax: +33 299 281458
E-mail: eric.feunteun@univ-rennes1.fr

Gelin, Curt
Scandinavian Silver Eel
P.O.Box 902
S-251 09 Helsingborg
Sweden
Tel.: +46 42 142433
Fax: +46 42 143176
E-mail: farming@silvereel.se

Jessop, Brian
Department of Fisheries and Oceans
Bedford Institute of Oceanography
P.O.Box 1006
Dartmouth, NS B2Y 4A2
Canada
Tel.: +1 902 4262158
Fax: +1 902 4266814
E-mail: jessopb@mar.dfo-mpo.gc.ca

Klein Breteler, Jan
Organization for the Improvement of
Inland Fisheries (OVB)
Buxtehudelaan 1
P.O.Box 433
NL-3430 AK Nieuwegein
The Netherlands
Tel.: +31 30 6058445
Fax: +31 30 6039874
E-mail: kb@ovb.nl

Knösche, Reiner
Institut für Binnenfischerei
Potsdam-Sacrow
Jägerhof
D-14476 Groß Glienicke
Germany
Tel.: +49 33201 40640
Fax: +49 33201 40630
E-mail: reiner.knoesche@ifb-potsdam.de

McKinnon, Lachlan
Marine and Freshwater Resources Institute
P.O.Box 114
Queenscliff, Victoria 3225
Australia
Tel.: +61 3 52580212
Fax: +61 3 52580270
E-mail: lachlan.mckinnon@nre.vic.goc.au

Meerburg, Dave
Department of Fisheries and Oceans
Fisheries Research Branch
200 Kent Street
Ottawa, Ont. K1A 0E6
Canada
Tel.: +1 613 9900286
Fax: +1 613 9540807
E-mail: meerburd@dfo-mpo.gc.ca

Müller, Rudolf
Chairman, EIFAC
EAWAG
Fisheries Section
CH-6047 Kastanienbaum
Switzerland
Tel.: +41 41 3492130
Fax: +41 41 3492162
E-mail: rudolf.mueller@eawag.ch

Naeve, Heiner
Secretary, EIFAC
Fishery Resources Division
Food and Agriculture Organization
of the United Nations
Viale delle Terme di Caracalla
1-00100 Rome
Italy
Tel.: +39 06 57056442
Fax: +39 06 57053020
E-mail: heiner.naeve@fao.org

Pedersen, Michael
Danmarks Fiskeriundersøgelser
Afdeling for Ferskvandsfiskeri
Veilsøvej 39
DK-8600 Silkeborg
Denmark
Tel.: +45 33 963128
Fax: +45 33 963150
E-mail: mip@dfu.min.dk

Rosell, Robert
Department of Agriculture and Rural
Development
Agiculture and Environmental Science
Division
Newforge Lane
Belfast BT9 5PX
United Kingdom
Tel.: +44 2891 255506
E-mail: robert.rosell@dardni.gov.uk

Russell, Ian
Centre for Environment, Fisheries and
Aquaculture Science (CEFAS)
Lowestoft Laboratory
Lowestoft, Suffolk NR33 0HT
United Kingdom
Tel.: +44 1502 524330
Fax: +44 1502 513865
E-mail: i.c.russell@cefas.co.uk

Sancho, Gorka
Technological Institute for
Fisheries and Food (AZTI)
Department of Fisheries Resources
Txatxarramendi Ugartea z/g
E-48395 Sukarrieta
Spain
Tel.: +34 94 6029400
Fax: +34 94 6870006
E-mail: gsancho@azti.es

Westerberg, Håkan
National Board of Fisheries
P.O.Box 423
S-40126 Göteborg
Sweden
Tel.: +46 31 7430333
Fax: +46 31 7430444
E-mail: hakan.westerberg@fiskeriverket.se

The European Inland Fisheries Advisory Commission/International Council for the Exploration of the Sea (EIFAC/ICES) Working Group on Eels met at ICES headquarters from 28 to 31 August 2001 to finish the work initiated at its 1999 meeting on defining biological reference points for European eel management use. The review of available information revealed that the European eel stock is in decline and that fisheries are outside safe biological limits. Anthropogenic factors (exploitation, habitat loss, increased predation, contamination and transfer of parasites and diseases) as well as natural processes (climate change) have contributed to the decline. Latest recruitment data (spring 2001) indicated a further deterioration of the status of the stock. As management at local level has failed to address the global decline of the stock, the implementation of an international stock recovery plan is of utmost urgency. The Working Group recommended that an international commission for the management of the European eel stock be formed to organize monitoring and research on eel stocks and fisheries and to serve as a clearing house for regular exchange of information regarding landings and resource status.

ISBN 92-5-104941-6 ISSN 0258-6096

9 789251 049419

TR/M/Y4676E/1/8.03/1500